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Integrated Modelling of the Energy, Water and Food Nexus to Enhance the Environmental Performance of Food Production Systems

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ABSTRACT

There is a need to explore ways in which current assessment tools can be enhanced to measure the true environmental impact of systems delivering products and services. The objective of this paper is to present an integrated energy, water and food (EWF) resource model that illustrates the interdependencies between different systems and sub-systems. The sub-systems considered in the environmental performance assessment of a proposed food production system in Qatar include: the food sub-system encompassing the production and application of fertilizers and the raising of livestock; the water sub-system which is represented by the reverse osmosis (RO) desalting process for water production and the energy sub-system, which considers power production utilizing combined cycle gas turbine (CCGT), solar Photovoltaics (PV) and biomass gasification integrated combined cycle (BIGCC). Life Cycle Assessment (LCA) forms the basis of the integrated EWF modelling tool and is used to evaluate the performance of the proposed food system.

Keywords: Energy, Water, Food, Nexus, LCA

1. Introduction

Food systems can no longer be analyzed as traditional linear systems. In fact, they have evolved into complex interconnected systems with social, economic and environmental dimensions. Every food system considers their product in terms of its utilization, availability and accessibility capacity, all of which are key components of food security defined as “when all people, at all times, have physical economic access to sufficient, safe and nutritious food to meet their dietary needs and food preferences for an active and healthy life” (FAO, 1996). Ericksen (2008) states that the characteristics of any given food system should be comprised of; the interaction between and within the bio-geophysical and human environments, the activities that take place within the food system and the outcomes which are directly related to the degree of attainable food security.

The challenge to overcome global food insecurity will only become more challenging as the world’s population increases to nine billion and food production capacity is challenged with increasing; land competition, resource scarcity and effects of climate change becoming apparent. It is imperative that food production systems develop in order to alleviate global hunger and to do so in a way that is environmentally and socially sustainable (Godfray et al., 2010). To this regard, it has been suggested the world will need to mobilize food systems to increase their total production capacity by 70 % to 100 % (World Bank, 2008). Although Godfray et al. (2010) stipulate that there is no simple solution to feed nine billion people, they do suggest a series of enabling strategies which need to be part of the overall mobilization initiative and they include; expanding aquaculture, closing the yield gap, increasing production limits, reducing waste and accommodating the changing diets.

The mobilization of food systems to meet the challenge of feeding the growing population will need to take the form of what is known as “sustainable intensification”. A strategy that is necessary in ensuring environmental integrity whilst increasing agricultural production (Godfray et al., 2010). Today, agriculture is the largest consumer of fresh water with a 70 % share of global water utilization in which the current population is appropriating 54 % of the accessible fresh water reserves. Furthermore, agriculture holds a 13.5 % share in global GHG emissions. In addition, there are upstream industrial activities that are of sole benefit to agricultural practices that need to be considered. The production of nitrogen based fertilizers which are equivalent to an additional 0.8 % of GHG emissions is one such activity (Brentrup and Pallière, 2008).

There is a need to adopt a systems approach when increasing agricultural production. For instance, Ericksen (2008) undertakes a holistic approach to food system design as illustrated in Figure 1. The system considers the drivers, interactions and feedback loops related to delivering food security. The objective of a systems approach is to identify key processes and determinants that influence outcomes within complex systems.

Figure 1.Food system components (Ericksen, 2008).

The framework considers the interactions between the human and natural environments in addition to whole chain inputs into the system with a food security focus for its outputs. Extending this framework based on the EWF nexus would strengthen the representation of natural resource inputs by considering their interdependencies and linkages with the environment. In this respect, the objective of this paper to present an assessment tool which can describe any system utilizing EWF resources in terms of its sub-systems. The approach required to develop such a model largely depends on the degree of resolution required. In addition, the nexus tool should be modular and so that complex systems are accurately represented allowing to identify inefficiencies easily. The modular nature of the tool ensures that the technical, spatial and temporal differences that exist between different systems and unit operation effects can be accounted for by modifying appropriate parameters of the component unit processes. The ultimate objective of the tool is to assist policy makers in answering key questions regarding the utilization of resources and the impact a particular policy may have on the environment. This paper will consider the fundamental question concerning the environmental burden in the provision of food in water-scarce countries such as Qatar. Although the cross-cutting tool integrates the utilization of EWF resources in one resource model and estimates the performance of any given system delivering a product or a service on atmospheric, terrestrial and marine eco-systems, this study only reports the impacts in terms of global warming potential (GWP). A study of the environmental burdens on terrestrial and marine eco-systems, including those related to marine eco-systems from desalination processes, are reported in earlier work by the authors (Al-Ansari et al., 2014).

Qatar is an arid country that suffers from a severe lack of natural water resources. Given that it is a small country home to approximately two million people, it possesses a disproportionate distribution of natural resources. This characteristic will affect Qatar’s ability to become fully self-sufficient, as with any nation, resulting in a degree of dependence on global trade to satisfy domestic food requirements. Qatar’s situation is particularly interesting; this is because whilst it has an abundance of energy reserves, it has a severe shortage of fresh water and arable land. It is also situated in a volatile region with extreme security risks and is part of a global community that is vulnerable to the effects of climate change, which will eventually exacerbate its natural disabilities. Annual freshwater extraction from aquifers is four times the rate of natural recharge of 50 Mm^3/y. The depletion is driven by agriculture which only accounts for 8 % of domestic consumption, and is leading to greater salination of aquifer water. Consequently, fossil fuel powered desalination provides more than 99 % of Qatar’s water demand providing up to 539 Mm^3/y. In 2012, Qatar’s electricity generating capacity reached 9,000 MW and is expected to rise with increasing population growth. The total arable land in the Gulf Cooperation Council (GCC) countries is in the order of 1.7 % and as a result imports at least 80 % of their food requirements. In a business as usual scenario food imports will continue to increase in proportion to population growth estimated at 14 %. The case study presented considers an expansion in Qatar’s domestic food system to a production a target of 40 % from the current 8 % using a specific crop profile by the year 2025 and a re-configuration through technology differentiation represented by the sub-systems.

2. Methods

The EWF nexus utilises Life Cycle Assessment (LCA) in order to compile the inputs and outputs for the food system and evaluate the potential environmental impacts in the delivery of its products throughout its life time.

The objective of LCA is to understand and evaluate the magnitude and significance of the potential environmental impacts of a product system. The four main stages of LCA according to ISO 14040 are: the Goal and Scope definition; the Life Cycle Inventory (LCI) analysis; the Impact Assessment (LCIA); and the Interpretation of the results. The Goal and Scope definition states the aim of an intended LCA study, the system boundary, the functional unit and the resolution (level of detail) in relation to this aim. The LCI analysis is the phase which quantifies the input/output relationships and compiles an inventory of input/output data for all component processes involved in the life cycle of the system under study. The nexus tool presented employs the CML 2001 baseline impact categories (including GWP), category indicators and characterisation methods. Inventory data are assigned to categories via the characterisation factors. In this particular case study, the nexus system functional unit is a particular crop profile measured in tonnes of output where the individual sub-systems are also considered in terms of their respective reference units and scaled accordingly, so as to serve the selected crop profile.

The model represented in this study is an extension of earlier work presented by the authors (Al-Ansari et al., 2014). The EWF nexus is defined as a series of sub-system LCI models that are developed to quantify material flows, natural resource and energy consumption at component unit process level. The LCI models are built using a combination of mass balance models, literature emission factors and engineering calculations which are validated using published literature and industry data. The flexible structure of the LCI database, provided through modularisation, enables the practitioner to choose component unit processes so that different technological options can be considered without the need for redesign or loss of information. The nexus modelling system presented here has adopted a food perspective with the objective of evaluating the environmental impact when increasing domestic food production. The rationale behind the crop profile choice, which includes greenhouse, open field vegetables and fruits, is not the subject of this study.

The analysis presented here is conducted for three modes of operation and three scenarios all of which serve to deliver the same functional unit and are evaluated in terms of their global warming potential. The baseline scenario uses fossil fuel to power the entire EWF nexus system using a CCGT. The second scenario integrates solar PV to power the RO desalination plants. The third scenario uses solar PV to power the RO desalination plants and fertilizer production facilities including their respective water requirement. The conventional mode of operation does not utilize manure for re-use within the system. The second mode integrates a BIGCC for power generation. The final operating mode utilizes the gasification by-product biochar for the enhancement of agricultural productivity in addition to the power generated from the BIGCC.

Energy requirements for pumping and distribution of water, food processing facilities, irrigation and administrative buildings and the embodied energy of equipment are not considered for simplicity. Emissions associated with the import of crops outside the crop profile consumed domestically are not considered. Land occupation for desalination facilities, power plants and fertilizer production facilities are not considered in land footprint calculations. Waste water treatment and reclaim is not considered in this study as it is assumed that in Qatar there are social barriers which prevent the use of treated water for food crops.

2.1. Energy sub-system

Energy is the main driver of the EWF nexus system. Three energy related sub-systems have been developed to power the EWF system. This section details the sub-systems which consider power generation using non-renewable natural gas fuel, renewable solar energy and the opportunity to generate energy from the gasification of the waste manure product from the food sub-system.

2.1.1. Combined cycle gas turbine

The combined cycle gas turbine (CCGT) model developed by Ibrahim and Rahman (2012) is based on a Brayton cycle based topping cycle and Rankine cycle bottoming cycle and is used to determine the power output of a turbine for any given composition of natural gas. The model considers the effect of operating parameters, such as peak pressure ratio, gas turbine peak temperature ratio, isentropic compressor efficiency and air fuel ratio, on the overall plant performance. The model uses a reference unit of 1 MWh, assumes a fuel to air equivalence ratio at 0.85 and calculates a total thermal efficiency in the range of 50-60 % in line with predicted values. Furthermore, the LCA model developed by Korre et al. (2012) which can be used to evaluate the performance of various CCGT power generation plant configurations has been integrated with the thermal efficiency calculations in order to complete the LCI database with a spectrum of emissions from power generation as illustrated on Figure 2.

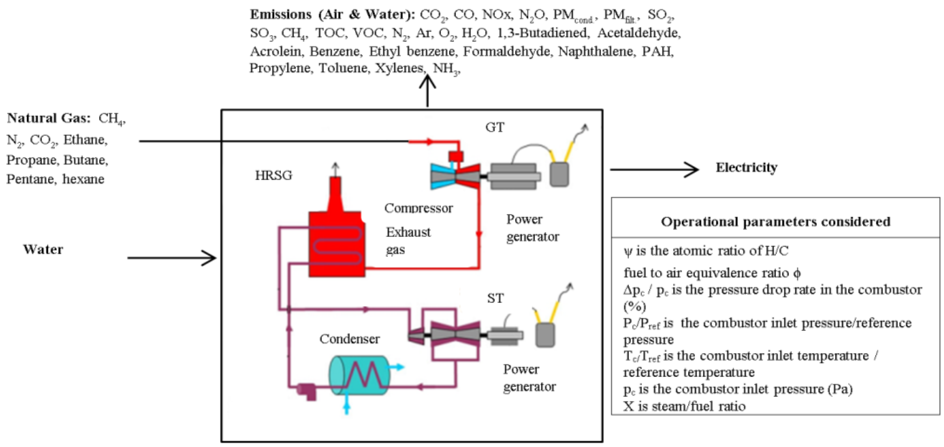


Figure 2. CCGT LCI model unit processes, inputs and outputs.

2.1.2. Solar photovoltaics

This study considers the procedure described by the RET screen model for the development of the solar PV sub-system (RETScreen, 2004). A 100 MW power plant utilizing monocrystalline-silicon modules is considered as the reference unit for this sub-system. The model calculates a daily global horizontal irradiance which is then then converted into an electricity generation potential using the PV module efficiency, the average module temperature, the nominal operating cell temperature, the clearness index and the area of the module. The life cycle emissions for solar PV which are released during other phases during the lifetime of the PV module (Fthenakis, 2011) are also considered. These include the fossil-fuel energy required to produce materials for solar cells, modules and the smelting, production and manufacturing facilities. It is assumed that PV modules are manufactured using power from a CCGT power plant and the natural gas consumption is aggregated over a 30 year life time.

2.2. Water sub-system

With increasing demand for water in isolation to power and improved anti-fouling/scaling membranes, the use of the RO desalting process is increasing. Furthermore, the specific energy requirement of RO systems is significantly lower than other desalination technology options resulting in a lower fuel requirement for a given desalting capacity. For instance thermal desalting systems such as Multi-Stage Flash (MSF) consume specific mechanical equivalent energy of about 4 kWh/m^3 of desalinated water, and heat energy in the range of 20 kWh/m^3, whilst RO desalting systems reduce the total energy requirement to 4-6 kWh/m^3 (Darwish and Al-Najem, 2000). Darwish et al. (2009) explored the true impact on the environment by considering the actual fuel consumption and the associated emissions from the provision of water from a variety of different power and water configurations. This study developed models for mechanically driven reverse osmosis desalting systems using the mechanical energy supplied from the CCGT and PV power plants. Water consumption in energy production is also considered. The water utilisation factors for a closed loop CCGT power plant and PV (negligible) used are based on Mielke et al. (2010).

2.2.1. Reverse osmosis

The RO model developed as part of this study (Figure 3) consists of a one stage pass system using an SWC 4+ membrane and calculates the power and fuel requirement per m^3 of desalinated seawater (Wilf and Awerbuch, 2007). The energy requirement is calculated using the net driving pressure (NDP). It is the driving force through a semi-permeable membrane which is defined as the fraction of the applied pressure in excess of the average osmotic pressure of the feed and any pressure losses within the system. The long term impact on the

Arabian Gulf, which is the body of water in which seawater is extracted from, can be found in Al-Ansari et al., (2014).

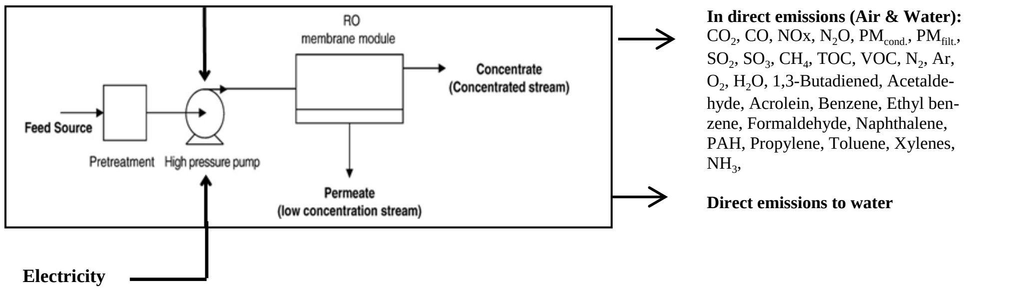


Figure 3. RO LCI model, inputs and outputs.

2.3. Food sub-system

The food nexus element encompasses fertilizer production and key agriculture practices such as the application of fertilizer and the raising of livestock.

2.3.1. Fertilizers

With respect to fertilizers, only the production and application of urea is considered in the quantification of emissions. Figure 4 illustrates the LCI model developed which integrates mass balance calculations with plant data and emission factors to calculate the emissions and resource consumption for ammonia and urea production. The electricity requirement encompasses the power required to drive the process and to convert water into steam.

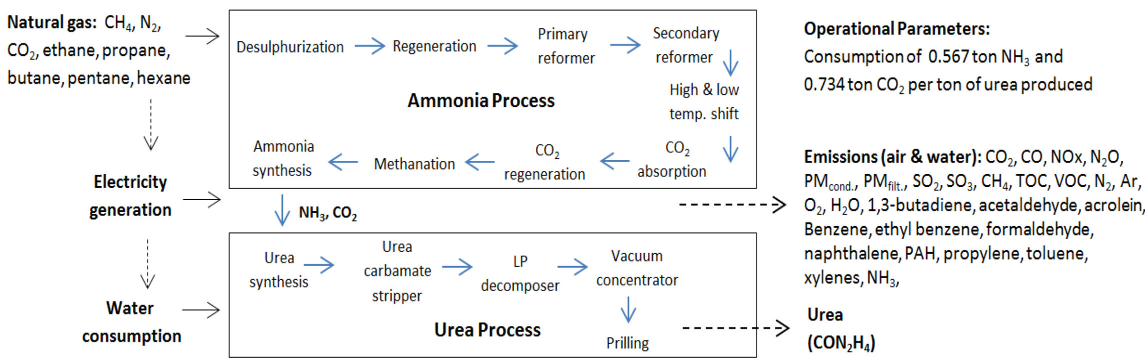


Figure 4. Ammonia and urea processes LCI model, inputs and outputs

2.3.2. Emissions from agriculture

It is estimated that the largest source of emissions from within the agriculture sector stem from crop and livestock production with an estimated 5 billion tonnes CO_2eq/yr. As such, the LCI developed for the agriculture sub-systems encompasses processes from both the raising of livestock and crop production (FAO, 2014).

2.3.2.1. Crop production

The emissions from crop production considered in this study include those from the application of urea fertilizer. Regarding the GHG emissions from fertilizer application, updated emission factors from the work of Brentrup and Pallière (2008) are utilised. The emissions include; urea hydrolysis (the release of CO_2 after application, equivalent to the CO_2 fixed during production), direct N_2O from use, indirect N_2O via NH_3, indirect N_2O via NO_3 and CO_2 from liming.

2.3.2.2. Livestock management

It has been estimated that livestock production represents approximately 18 % of the world's anthropogenic GHG emissions (Weiss and Leip, 2012). Furthermore, wastes from animal production systems contribute as much as 30 – 50 % of the global N_2O emissions from agriculture. Methane and nitrous oxide (CH_4, N_2O) emissions from manure management both contribute to global warming, whilst volatilization of ammonia (NH_3) contributes to acidification. The quantification of the GHG fluxes for all emission sources follow the IPCC (2006) tier 1 guidelines which include; methane emissions from enteric fermentation and manure management, direct and indirect N_2O emissions from manure management. The study assumes a scenario in which the manure is stored rather than applied on land.

2.3.3. Waste management

The environmental impacts from manure can be reduced if the manure is recycled into useful products that are utilised within the system. This paper considers the gasification of manure as the waste management option of choice. Gasification is a process in which organic matter (manure) is decomposed into syngas which can be used in power generation or converted into high value products (Cantrell et al., 2007). A mass balance based fixed bed gasification model using air and steam as the gasifying medium was developed (Thanapal et al., 2012; Gordillo and Annamalai, 2010). The process is governed by a series of primary heterogeneous chemical reactions (carbon-steam, Boudouard, hydrogasification, water-gas shift, methanation) which are both endothermic and exothermic and occurring at different rates (Priyadarsan et al., 2004, Gordillo and Annamalai, 2010). The operational parameters considered in the model include the equivalence ratio (ER), steam to fuel ratio (S/F) and feed composition. The model assumes that dairy manure is a representative composition for all livestock manure (Gordillo and Annamalai, 2010). The energy required for pre-gasification, moisture vaporization and the separation of nitrogen using an air separation unit (ASU) is considered in the input. Furthermore, this study assumes that the manure is collected every two months reducing the manure emissions by a factor of six in line with the LCA study developed conducted by Wu et al. (2013). Emissions from the collection of manure, its transport to and from industrial sites and the handling equipment are not considered.

Biochar is a nutrient rich by product which has the potential to reduce GHG emissions through carbon sequestration, N_2O emission reduction when applied soil, displacement of commercial fertilizer and enhancement of the agronomic efficiency. Uzoma et al. (2011) in a study regarding the effect of biochar on the productivity on sandy soils confirmed that the use of biochar improves the physio-chemical properties of coarse soil, nutrient uptake potential of the crop and its water use efficiency (WUE). The study concludes that application of biochar at mixing rates of 10, 15 and 20 tonnes per hectare increased WUE by 6 %, 139 % and 91 % respectively. Furthermore the study demonstrated that the application of biochar affects the plant nitrogen (N) uptake in addition to the other nutrients. Increasing plant N uptake through biochar amendment improves N fertilizer use efficiency especially in sandy soils which is the prevalent soil type in Qatar. At the optimum application rate of 15 kg/ha the N uptake increased by approximately 53 %. For this study, the effects of a conservative 50 % water and fertilizer reduction on the GWP through the application of biochar at rates of 15 kg/ha are examined. The quantity of biochar produced is assumed to be 20 % of dry matter in the manure Wu et al. (2013).

2.3.3.1. Biomass Integrated Gasification Combined Cycle (BIGCC):

The gasification process produces a low calorific syngas which can be used in traditional combined cycle gas turbines configurations. Whilst it is possible to design and build greenfield CCGT systems based on syngas fuels, it is also possible to co-fire gasified product with natural gas (Rodrigues et al., 2003) and modify existing facilities to utilize syngas as the fuel for power generation (Chacartegui et al., 2013, Kim et al., 2011). The combined cycle gas turbine sub-system described in section 2.1.1 is thermodynamically reconfigured to utilize the syngas from the gasified manure by changing turbine operating temperatures in addition to incorporating the syngas properties including composition with a corresponding heating value. Figure 5 illustrates the coupling of a dryer, ASU, gasification unit and a CCGT to form a BIGCC system. The BIGCC operates with an efficiency of approximately 35 % with a gasification temperature of 800 °C. The efficiency takes into account the energy for pre-gasification and the energy input into the ASU. The syngas produced contains a heating value of 9,100 kj/kg. In practical applications mechanical modifications would need to be made to the combustion chamber, gas turbine and the bottoming cycle (Chacartegui et al., 2013, Kim et al., 2011). Air emissions were based on mass balance calculations and emission factors (Mann and Spath, 1997). Energy requirement for the generation of steam has been accounted for. Furthermore it is assumed that auxiliary systems such as the dryer and ASU utilize power from within the system and not from an external source.

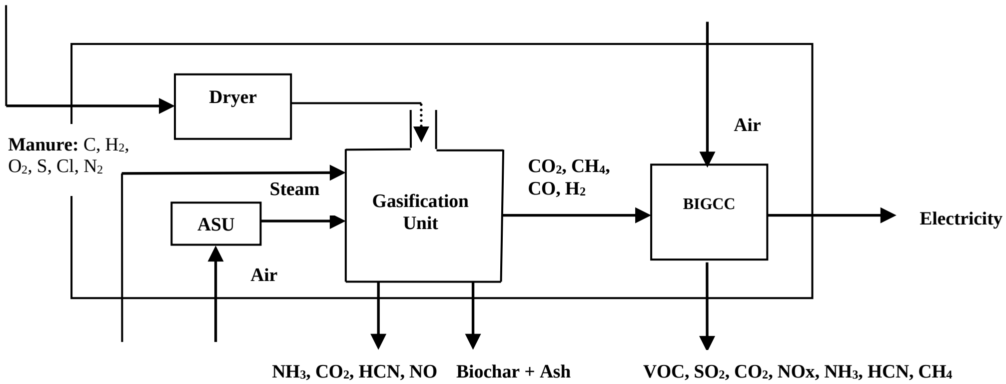


Figure 5. Gasification of manure for power generation system diagram.

3. Results and discussion

This study explored three fixed scenarios for the nexus configuration delivering a certain food production profile for the state of Qatar under three operating modes. It is important to note that the energy saving potential from the BIGCC is evenly distributed amongst all energy using sub-systems within the nexus rather than as a full allocation to one individual sub-system. Furthermore, the same assumption applied to the livestock management sub-system with respect to carbon neutrality is applied to the BIGCC. Therefore, CO_2 emissions released from the BIGCC are not considered in lifecycle emissions as it is considered to have been extracted from the atmosphere. Furthermore, in the LCA conducted by Mann and Spath (1997) on biomass power it was concluded that over 90 % of N_2O emissions in the gasification process was a result of livestock management which was considered as part of this study. As such, it is assumed that N_2O emissions from the gasification process are nill.

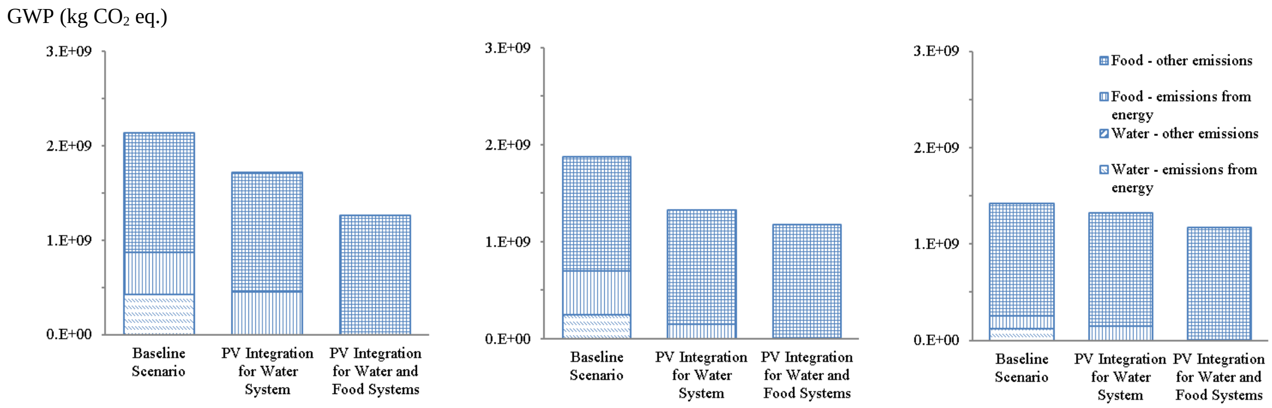
The main performance parameters from the BIGCC are illustrated on Table 1. The recycling of manure has the capacity to provide 30 – 40 % of the nexus energy requirements resulting in a natural gas saving of approximately 1.5×10^5 tonnes annually in terms of energy equivalence.

Table 1. Gasification performance parameters.

Gasification performance	Value
BIGCC operating efficiency	~ 35 %
Nexus energy substitution potential	~ 30 - 40 %
Natural gas equivalent	~ 1.5×10^5 kg/yr.
Total biochar production	~ 400,000 tonnes/yr.
CO_2 emission rate (calculated)	~ 900 kg/MWh
CH_4 emission rate (Mann and Spath, 1997).	0.00027 kg/MWh

Furthermore, the generation of biochar is significantly large in the range of 400,000 tonnes annually. Although research on the effects of biochar is ongoing, it is considered to be essential in soil regeneration especially in arid lands such as Qatar. The consequence is a significant increase in nutrient uptake and water use efficiency.

Figure 6 shows that the largest sources of GHG emissions originate from the food sub-system for all scenarios and operating modes. The source of which are the non-energy related emissions from the livestock sub-system which releases large amounts of CH_4 and N_2O, both of which have high GWP's. It is also clear that the total GWP for all three operating scenarios decrease with the introduction of the BIGCC. A comparison of the conventional nexus configuration with the BIGCC integrated system, shows a reduction in GWP for each of the three scenarios; baseline (20 %), PV integrated in the water system (45 %) and PV integrated for the water and food system (52 %). The total reduction of GWP with further utilization of the biochar is in the range of 33 %, 55 % and 62 % respectively in comparison to the conventional configuration.

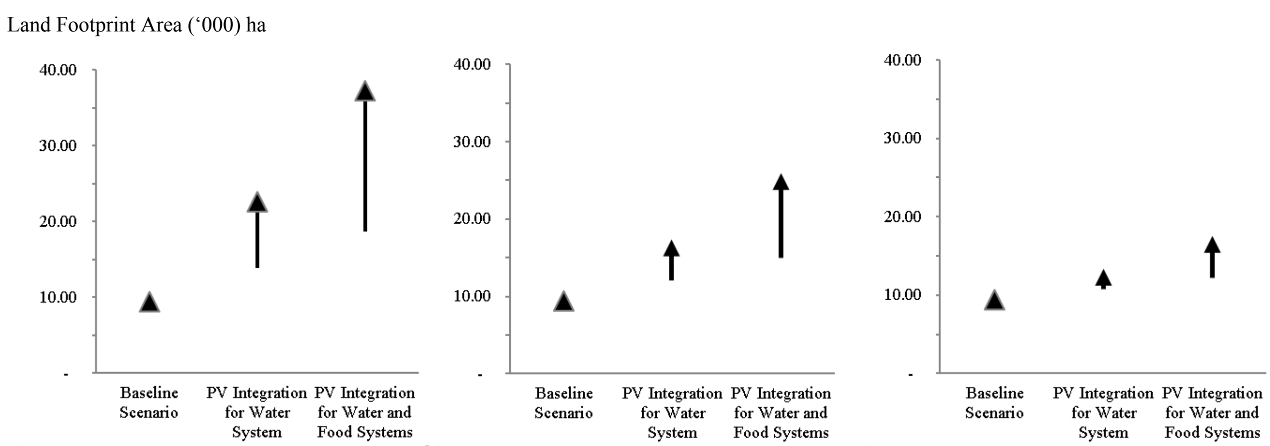


a) b) c)

Figure 6. Total GWP (kg CO_2 eq.) for the food production system for the three scenarios operating under different configurations a) conventional mode, b) BIGCC integration and c) BIGCC and biochar integration.

Evidently there is small improvement with the integration of biochar. This is because it is assumed that the biochar had no impact on the livestock management sub-system which is the largest emitter of GHG's. However, it is important to note that this study adopted a conservative 50 % improvement in WUE and 50 % improvement in nutrient uptake, whereas previous studies have recorded higher improvement. It is considered that further research is necessary to accurately quantify the improvements of biochar on the health of the soil, WUE and nutrient uptake.

The land footprint requirement rises with increased reliance on PV to power the nexus as illustrated on Figure 7. The largest PV land footprint is the upper limit of the conventional nexus configuration with a full PV deployment to power water and food systems requiring a total investment of 37,000 ha. This can be reduced by over 40 % with the integration of the BIGCC and the utilization of biochar. Within the nexus, natural gas is utilized in the CCGT and for ammonia production. In the conventional mode, the natural gas consumption required for the production of ammonia is insignificant in relation to the power generation. In the full deployment of PV to power the water and food system scenario, the natural gas consumption required to manufacture the PV aggregated over 30 years remains larger than the annual ammonia requirement with a 97 % of the total natural gas consumption.



a) b) c)

Figure 7. Total land footprint (thousand ha) requirement for the three scenarios operating under different configurations; a) conventional mode, b) BIGCC integration and c) BIGCC and biochar integration.

For the sub-systems considered, the integration of the BIGCC reduces the natural consumption required for energy amongst the three scenarios as illustrated on Figure 8. In fact, Figures 8b and 8c illustrate the natural gas credit which can be generated with the integration of PV and BIGCC with a further credit achievable with the integration of biochar.

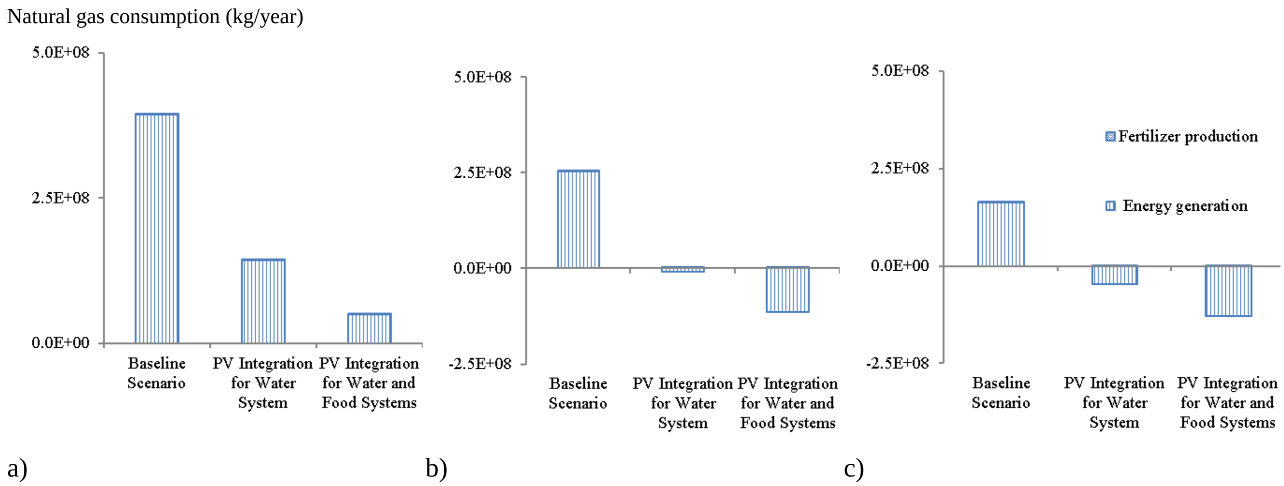


Figure 8. Total natural gas requirement (kg/year) for the three scenarios operating under different configurations; a) conventional mode, b) BIGCC integration and c) BIGCC and biochar integration.

4. Conclusions

The results of this study indicate that the BIGCC is a viable option to reduce environmental degradation with the quantity of waste generated in the livestock system. Further research is required to model the carbon fluxes within the system in order to accurately account for carbon emissions from the food sub-system and to the extent carbon neutrality is actually achieved. Furthermore, field scale experimental research is required in order to understand the true potential for agricultural productivity enhancement through the use of biochar. It is possible that this study may have underestimated the potential environmental benefits given the conservative 50 % improvement rate for WUE and nutrient uptake used in the analysis. The objective of such research would be to identify the optimum productivity increase that would need to be achieved in order to balance out the GHG emissions from the livestock sector. Additional onsite experimental research is necessary to gather the data required to adopt IPCC 2 and 3 analysis for livestock emissions as this study is based on the tier 1 approximation.

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Using of the LCA methodological framework in perennial crops: Comparison of two contrasted European apple orchards

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ABSTRACT

Life Cycle Assessment (LCA) is a helpful approach for a better understanding of the environmental impacts of apple orchard cropping systems. The application of LCA to perennial cropping systems raises methodological questions such as the consideration of non-productive stages. Two recent reviews give recommendations (Cerruti et al., 2011) and a methodological framework to deal with perennial crop cycles (Bessou et al., 2012). The objectives of our study were to test these recommendations and framework by comparing two contrasted apple fruit production systems (intensive and semi-extensive) and to assess the weight of the non-productive stages in the orchard life cycle impacts. Unproductive stages weighted up to 21% and 28% of the studied impact categories, in the semi-extensive and intensive orchard respectively, with little contribution of the nursery stage. The consideration of the unproductive stages is discussed and the necessity to explicit the approach used to account for the duration of perennial cropping systems is also outlined.

Keywords: LCA methodology, perennial crop, apple production, orchard conservative design, orchard management

1. Introduction

Life cycle assessment (LCA) is a comprehensive methodology, which considers the whole life cycle of the cropping systems. In the case of perennial crops such as orchards, unproductive then productive stages characterize the cropping system. This aspect raises methodological questions about the way to account for all orchard stages in the total impact assessment. Two recent reviews (Cerutti et al., 2011; Bessou et al., 2012) recommend integrating those non-productive stages in the impact assessment. Moreover Bessou et al. (2012) propose practical guidelines for perennial cropping systems according to data availability to describe and account for those stages in a LCA study. These guidelines propose to encompass every stage of the perennial cropping cycle, as well as different approaches to model it. Our main goal was to test the methodological recommendations of these authors, by analyzing the contribution of each life cycle stage in the total environmental impact of two contrasted apple orchard systems: one intensive and one semi-extensive. Intensive systems refer to orchards managed to maximize fruit production, usually including several of the following design traits and management practices: dense planting of short-lived trees on dwarfing rootstocks, high chemical inputs, intensive pruning to shape the trees in a restricted form, and frequent mowing the orchard groundcover (Dart, 2008). In contrast, orchards that are managed extensively request less use of pesticides and fertilizers with relatively long-lived trees that could reach the veteran stage. We expected these two orchard systems to affect the relative importance of the non-productive stages in the LCA and therefore, to be good case studies to use with this newly developed guidelines.

2. Methods

Two existing and contrasted apple orchards were compared following the methodological guidelines proposed by Bessou et al. (2012) and Cerutti et al. (2011) to analyze perennial cropping systems with LCA. To comply with current methodological frameworks for perennial crop LCA, the different stages of the orchard life cycle including non-productive stages, were accounted for. The non-productive stages included nursery, orchard creation (planting), orchard establishment and destruction.

2.1. The studied apple cropping systems and their modelling

The two studied orchard systems were described and modelled from on-farm surveys. The geographical origin, namely Northern (Picardy) and Southern (Rhone Valley) France, for the intensive and semi-extensive systems respectively, ensured the data consistency of each modelled apple production system (Bessou et al., 2012). The main characteristics of the studied orchards are summarized in Table 1. The two studied orchards differed in their lifetime, orchard height and associated machinery use, irrigation management (in relation with the climatic context) and planting distances. Moreover, the intensive orchard establishment stage lasted two years compared to one year in the semi-extensive orchard. Indeed, the semi-extensive orchard was harvested as soon as the annual production reaches around 3 to 4 tons/ha, i.e. in its second year after planting. In the intensive orchard, fruits were harvested only once the production has reached a yield of 20 tons/ha (namely in the 3rd year), in order to optimize the costs.

Table 1. Main characteristics of the two studied production systems.

Characteristics	Intensive orchard	Semi-extensive orchard
Cultivar	Jonagold	Golden
Planting density (number of trees/ha)	2500	1100
Tree height (m)	2.5	4.5
Pruning	Mechanical	Manual
Between-row management	Mowing and mulching	Mowing and mulching
Irrigation system	No irrigation	Buried irrigation system with localized drippers
Weed, pest and disease control	Mainly chemical	Mainly chemical
Harvest method	Manual	Manual with elevator
Lifespan (years):	15	26
- Orchard establishment	Year 1 and 2 (2 years)	Year 1 (1 year)
- Productive stage (commercialized apples)	Year 3 to 15 (13 years)	Year 2 to 26 (25 years)
Annual average yield over the productive stage (t/ha/year)	55.4	37.8
Cumulated commercialized yield over orchard whole life time (t/ha)	720	944.7

For both orchards, all cultural practices related to fertilization, plant protection, between-row management, tree training, fruit load management as well as harvest were collected. As recommended by the two reviews, the modelling of each orchard stage, namely the unproductive stages (i.e., nursery, orchard creation, establishment and destruction) and the productive stage (i.e., years with apple commercialization) was included in the analysis. During the nursery stage, grafted trees were produced, which were considered as inputs in the orchard creation stage. The orchard creation stage corresponded to the soil preparation before planting, planting and installation of orchard infrastructure such as poles, wires and the irrigation system if the orchard is watered. During the orchard establishment stage, fertilizer doses corresponded to one third of those of the full production years. The inputs of an average production year are listed in Table 2. The two studied orchards differed mainly in the fertilizer and pesticide doses, which were higher in the intensive orchard. The last stage of orchard destruction entailed the removal of trees and infrastructure.

Table 2. Input list for one average productive year for the two production systems, with N: Nitrogen and a.i.: active ingredient.

Input	Intensive orchard	Semi-extensive orchard
N fertilizer rate (kg N/ha)	114	47.3
Pesticide, active ingredient (kg a.i./ha)	74	28.5
Including copper and sulfur (kg a.i./ha)	44	15
Fuel consumption (L/ha), including harvest	289	234
Including self-driven elevator fuel consumption (L/ha)	-	67
Pesticide and fertilizer and growth regulator applications (runs/ha)	47.4	41
Other mechanical practices (pruning, mowing, mulching) (runs/ha)	5.7	4

Following Bessou et al. (2012), the data available for the two orchards implied to use two different approaches to describe the studied orchards during their whole lifetime. Indeed, a modular approach, with each stage independently modeled, was used for the intensive system: data were recorded for the different stages from one single year. A chronological approach, which consists in describing all the historical course of the crop development, was used for the first eight years of the semi-extensive system, while neighbor orchards were the basis to estimate the characteristics of the following seventeen years.

2.2. System boundaries and functional units

The two studied systems encompassed life cycle phases from cradle (namely the production of the inputs of all the modelled stages), to the gate of the apple storage place. To compare the two studied systems and follow the recommendations of Cerutti et al. (2011), two functional units (FU) were used. The mass-based FU was calculated for 1 ton of apples for the cumulated yield over the whole orchard lifetime. The area-related FU was 1 $ha^{-1}.year^{-1}$ of land used to produce apples over the whole orchard lifetime.

2.3. Inventory and characterization methods

The inventories needed for the manufacturing and supply of inputs and buildings were taken from Ecoinvent V2.2. Following the recommendations of Cerutti et al. (2011), a nitrogen balance based on the tree requirements was calculated for each year of the establishment and productive stage of the orchard.

Climate change, terrestrial acidification, freshwater and marine eutrophications and energy consumption were calculated with SimaPro V7.3.3 software using Recipe method (Goedkoop et al., 2009) and cumulative energy demand method V1.08 (Althaus et al., 2007). Despite the importance of pesticide applications (Table 2), emissions related to the use of sulfur- and copper-based pesticides could not be computed with the existing fate models (Villanueva-Rey et al., 2014). As a consequence, ecotoxicity and toxicity impacts were not assessed in the present study.

3. Results

3.1. Contribution of the non-productive stages to the total environmental impacts

The relative contribution of each stage (non-productive and productive) to the five calculated environmental impact categories is presented for both semi-extensive and intensive orchard systems (Figure 1). Whatever the impact category and the FU, the unproductive stages represented up to 21 % and 28 % in the semi-extensive and

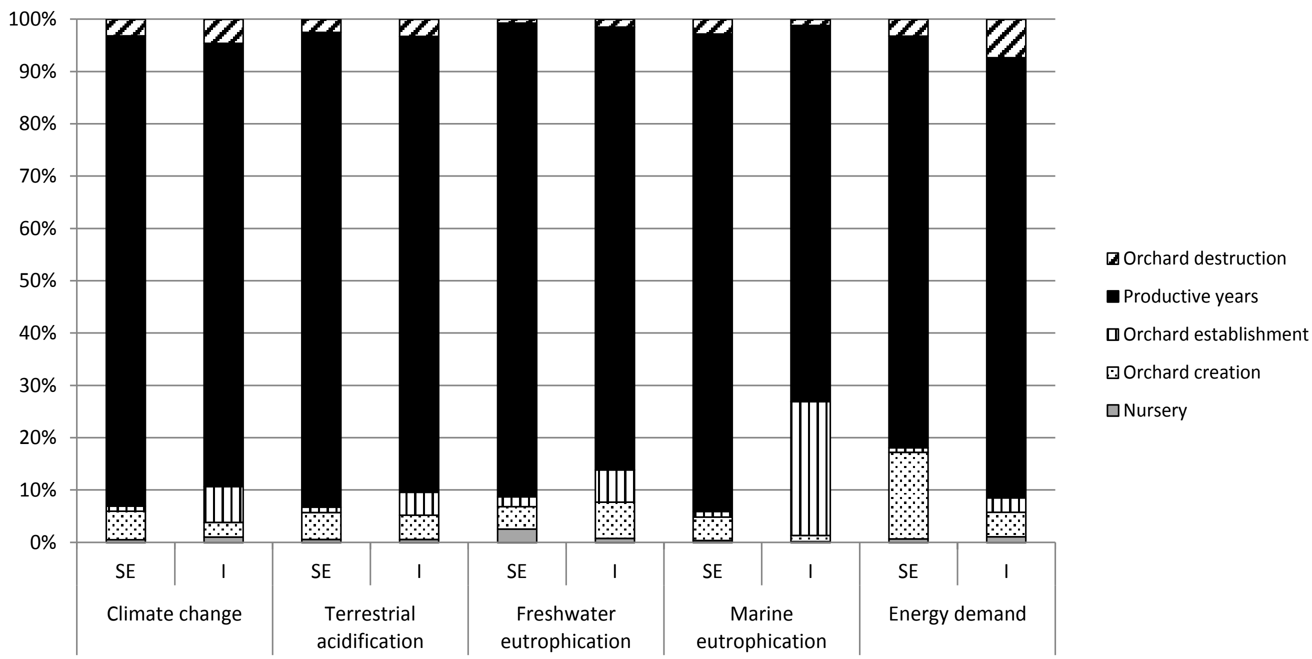

Figure 1. Relative contribution of each stage (productive years and non-productive stages) to five environmental impact categories in the semi-extensive (SE) and the intensive (I) orchard systems.

intensive orchards, respectively. The nursery itself accounted at maximum for 1% and 2.6% of the orchard whole life impact in the intensive and the semi-extensive orchards, respectively.

In the intensive orchard, the main contributor to climate change, terrestrial acidification, freshwater and marine eutrophication impacts among non-productive stages was the ‘orchard establishment’, while the ‘orchard creation’ stage accounted for 3% to 7% for climate change, terrestrial acidification and freshwater eutrophication impacts. In the semi-extensive orchard system, the main contributor to all calculated impact categories among the unproductive stages was the ‘orchard creation’ (i.e. soil preparation, planting and orchard infrastructure creation).

These differences among orchards can be explained by the duration of the ‘orchard establishment’ stage and the climatic constraint. Indeed, the establishment phase of the intensive orchard entailed two unproductive years, versus one in the semi-extensive orchard. Besides, as there was no irrigation system in the intensive orchard due to sufficient rainfall to fulfill the crop requirements, the relative weight of the creation stage was lowered.

The contribution of the non-productive stages to the total impacts was lower in the semi-extensive orchard compared to the intensive orchard for all the studied impact categories, except for energy demand with similar or higher value according to the FU. This result could be related to the energy consumption occurring during the ‘orchard creation’ in the semi-extensive orchard. Indeed, planting was mechanized and required heavy equipment, whereas it was manual in the intensive system. Moreover, the semi-extensive system was irrigated with a buried irrigation system, which also required heavy machinery use to install it. The energy demand for the non-productive stages mainly corresponded to the tree removal (orchard destruction) in the intensive orchard.

3.2. Environmental impact of the two orchard systems

The impact per (ha.year) of the intensive orchard was 1.5 to 11 times higher than the one of the semi-extensive orchard for climate change, terrestrial acidification, freshwater and marine eutrophication, except for energy demand with similar values. For the mass-based FU, the differences were less important, or opposite for freshwater eutrophication and energy demand (Table 3). Indeed the cumulated yield for both orchard types is relatively similar with 720 tons for the intensive orchard and 945 tons for the semi-extensive orchard, even though their lifetimes are different with 15 years for the intensive and 26 years for the semi-extensive orchard.

Table 3. Environmental impacts of the orchard systems as expressed per ton of fresh fruits and per (ha.year).

		ton^{-1}		**(ha.year)$^{-1}$**	
Impact categories	**Unit**	**Semi-extensive**	**Intensive**	**Semi-extensive**	**Intensive**
Climate change	kg CO_2 eq	75.24	89.79	2733.71	4309.94
Terrestrial acidification	kg SO_2 eq	0.710	0.985	25.789	47.269
Freshwater eutrophication	kg P eq	0.033	0.024	1.22	2.000
Marine eutrophication	kg N eq	0.037	0.155	1.33	15.005
Energy demand	MJ	1160	880	42143	42243

The higher environmental impact observed for most impact categories (except energy demand) in the intensive orchard was mainly related to fertilization, among which nitrogen applications. Indeed the production and in-field emissions of fertilizer inputs contributed to 55% and 28% of the climate change in the intensive and semi-extensive orchards, respectively. Terrestrial acidification and marine eutrophication were mainly explained by nitrogen fertilization in the intensive orchard (70 to 86% of the impact), whereas it was related to both fertilizers and pesticides production and use (50% of the total impact) in the semi-extensive orchard.

Regarding the energy demand, the mean annual fuel consumption during the productive years was lower in the semi-extensive than in the intensive orchard, but it was balanced by the mechanization of the semi-extensive system, which was higher over the whole orchard life cycle compared to the intensive orchard. Indeed, the machinery and the infrastructures used to plant and irrigate the orchard were more important and more energy-consuming in the semi-extensive than in the intensive orchard. Moreover, due to the tree height in the semi-extensive orchard, a self-driven elevator was used for each manual operation (pruning, thinning and harvest).

4. Discussion and conclusion

The present study confirms the environmental burden of unproductive stages with up to 28% of the total impact, although the nursery itself was of little importance (1 to 2.2% of the orchard whole life impact).

It is noticeable that the present comprehensive results about climate change impact and energy demand are similar to those published by Mouron et al. (2006) for the area-related FU, while they are two to threefold higher than those published by Milà i Canals et al. (2006) and Alaphilippe et al. (2013) on similar semi-extensive orchards. Only Mouron et al. (2006) included the unproductive and establishment stages in its calculation, which explain that the results are comparable to the present study. The use in the present work of a chronological approach, with the consideration of several years of full production, with annual adjustment in crop operations and alternate fruit bearing also contributes to explain the differences among studies.

Our work outlines the necessity of standardization to model the perennial crop life cycle and attests that the methodological frameworks proposed by Bessou et al. (2012), and Cerutti et al. (2011) are relevant to assess global environmental impacts in orchards.

In conclusion, aside from marine eutrophication and cumulative energy demand, the weight of the unproductive stages in all impact categories was only slightly changed between our two contrasted orchard designs and managements. An intensive establishment stage and a semi-extensive creation stage may noticeably contribute to marine eutrophication and cumulative energy demand, depending on their durations and intensification levels. So, based on our results, we recommend that, when identifying the hotspots of apple production systems, LCA has to focus on and privilege an accurate representation of the field stages, including the orchard establishment and the orchard creation.

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Developing a model of Sustainable Production and Consumption of Cantabrian Anchovies: a case study of Life Cycle Management in the Fish Canning Industry

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ABSTRACT

The aim of this work is to develop an organization, management and marketing model of Cantabria anchovy with environmental considerations in order to obtain environmental sustainability by improving the efficiency of use of raw materials, waste reduction and recycling, utilization and enhancement of products, energy efficiency and carbon and water footprint. This production and consumption model contribute to the Spanish Challenges according to the competitiveness of the fisheries sector in national and international markets.
The management model of Cantabrian anchovies allows to inform consumers about the environmental impacts associated with anchovy canning industry by means of Eco labels. In addition, the model allows transmission of knowledge about environmental and socio-economic impacts related to the life cycle of the product to those who make decisions on production, consumption and waste management product, in order to promote better management lifecycle for a sustainable production and consumption.

Keywords: Anchovy, Life Cycle Assessment, Canning Industry, Sustainable Production and Consumption

1. Introduction

The canning industry generates a lot of wastes. In this context, industrial waste management in the food industry is an important environmental problem that requires specific action draft through a comprehensive and coherent policy on the prevention and control of wastes. In addition, the European Thematic Strategy (SEC 2011 70 final) on the Prevention and Recycling of Waste and its relation to the life cycle approach, the Directive 2008/1/EC on Integrated Prevention and Pollution Control (IPPC), the Integrated Product Policy (IPP) (SEC (2009)1707 final) and the Strategy on the Sustainable Use of Natural Resources can help promote more sustainable production and consumption models based on the life cycle approach and the participation of stakeholders.

The fishing industry for Cantabrian anchovy (Engraulis Encrasicolus) is one of the main economic resources of the fleet of the Cantabria Region. According to CONSESA (Association of Manufacturers of Canned Fish of Cantabria Region), in 2,012 there was a production of 13,267 tons of canned anchovy, equivalent to more than 91 million euros.

The Anchovy caught in the Cantabrian Region is intended for direct human consumption either in the form of fresh fish, or to production of canned anchovies in oil. The usable of the fish for food is between 60% and approximately 70%. An important amount of waste streams occurs mainly in operations heading, gutting and packing. Taking into account that 40% of the weight anchovy captured just as waste, it is estimated that nearly 9,000 tons of this resource are wasted and also a lot of waste effluents (mainly water and oils) are obtained.

According to this, it is necessary a sustainable Cantabrian anchovy industry taking into account local considerations for global development. In this sense, it is necessary to design and implement strategies for sustainable management of Cantabria anchovy industry under a life cycle approach. These strategies will be focused mainly on increasing the utilization of wastes from anchovy production to obtain co-products with higher added value that can be allocated to new green markets. This new development is based on the application of tested methodologies (Life Cycle Analysis, Best Available Techniques and Eco-labels).

The canning industry uses a variety of types of packaging of different sizes and materials. Packaging the final product requires metallic packaging or glass packaging, with or without secondary packaging. After consuming the product, the packaging (which contain one part oil) should be managed appropriately. Directive 94/62/EC aims to limit the production of packaging waste and promote recycling, reuse and other forms of recovery of waste. In this context, it is very important the prevention and proper treatment of all waste generated in the lifecycle of the canned anchovy, including packaging.

The European Commission Joint Research Center developed a new method to calculate the environmental performance of a product based on LCA methodology. It has also developed the Single Market for Green Products to ensure that consumers are not confused by the environmental unclear information. In this sense, the presented methodology has as one of its main objectives developing a Product Category Rules (PCR) for the creation of an Environmental Product Declaration (EPD), which establishes the principles for communicating the environmental performance, such as transparency, reliability, completeness, comparability and clarity of the product. According to this, it is possible to promote the supply and demand for more sustainable products.

2. Proposed Model

The mode will be based on the concept of "with science and society". The methodology will be focused on the needs of society. In this sense, will be consider to governments and their representatives, businesses, social organizations and citizens. Figure 2 shows the diagram of the proposed model and the Life Cycle Approach on the proposed model.

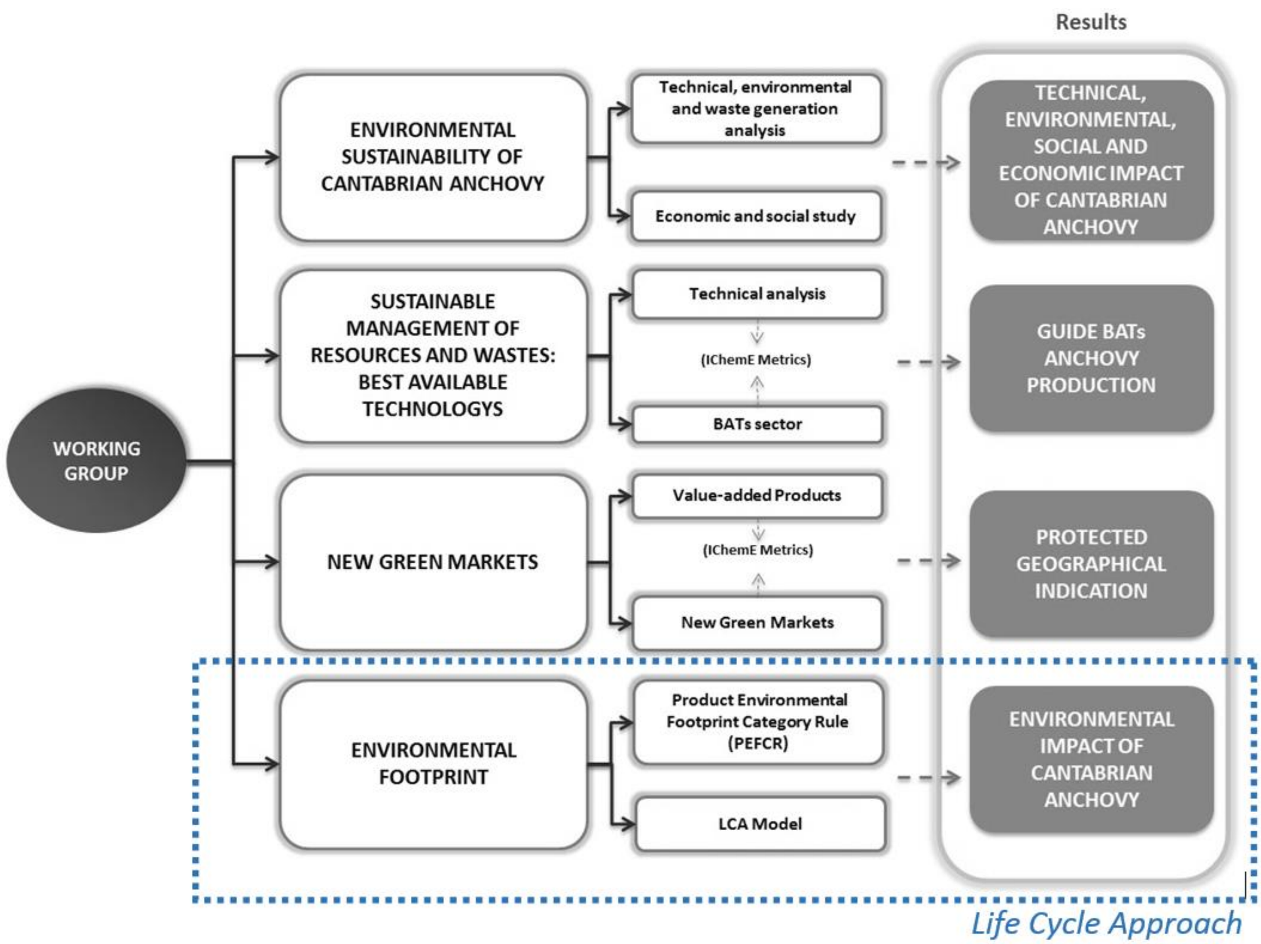

Figure 2. Diagram of the work methodology. Life Cycle Approach on the proposed model.

The main objective of the proposed model is to design and implement strategies for sustainable management sector of the Cantabrian anchovy under a vision of the life cycle. These strategies will mainly focus on increasing the use of generated waste effluents, in order to obtain co-products with higher added value that can be allocated to new green markets. The model will base on the application of tested methodologies (Life Cycle Analysis, Best Available Techniques, Ecodesign and Environmental Footprints) (Avadí and Fréon 2013; Ayer et all. 2007; Azapagic 1999; Pelletier et all. 2007). On the other hand, the methodology will base on the opinions and experience of industry representatives. This working method aims to apply the principles of the Integrated Product Policy (IPP) and the Politics of Sustainable Consumption and Production of the European Union, so that the conclusions reached in the process can be useful in a possible process to review these policies, both nationally and internationally.

The main objectives are:

- Contribute to the sustainable development of the canning industry of Cantabrian anchovy, by promoting the supply and demand for products that respect the environment throughout its life cycle.
- Promote the competitiveness of Cantabrian anchovy by promoting the efficient use of natural resources and exploitation of waste streams and using appropriate recovery systems for subvalue-added-products. This competition necessarily involves the application of Best Available Techniques in the fishing and processing of anchovy and proposed solutions to overcome barriers to implementation. The BAT's are necessary to evaluate in order to assess environmental, economic and social impacts. This work considered the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE) that give a balanced view of the environmental impact of inputs-resource usage and outputs-emissions, effluents, and waste (Tallis et al. 2002).
- Disseminate to stakeholders (companies, government agencies, consumer organizations) valuable and updated information about the environmental impact and socio-economic development of fisheries and canned anchovy. In particular inform consumers about the environmental impacts associated with anchovy canning industry and its derivatives through the use of reliable eco-labels based on the methodology of Life Cycle Analysis (LCA).
- Optimize the final product packaging for marketing, analyzing the transformation process of the anchovy (sewage, oils, packaging waste and fish waste). It is necessary to develop the Product Environmental Footprint Category Rule (PEFCR) for the sector of anchovy, which would be used later to help consumers distinguish seafood and more respectful canners with the medium environment throughout their life cycle. Similarly, the development and dissemination of Footprint pursued.
- Monitoring and implementation of environmental legislation within a local and regional basis.

To achieve these broad objectives, will be held the following specific objectives:

- Coordination and Consultation Sector, including stakeholders.
- Environmental, economic and social Life Cycle Assessment of Cantabria anchovies. Updated and detailed analysis of environmental, social and economic sector.
- Sustainable management of resources and effluent canning Cantabrian anchovy: Analysis and Application of Best Available Techniques (BATs)
- New markets for canning Cantabrian anchovy.
- Environmental Footprint of Cantabrian anchovy.

3. Life Cycle Approach of the Proposed Model: Combining Type I and Type III Eco-Labels

Eco-labels are used by manufacturers and distributors to provide information about the environmental performance of their goods on a voluntary basis. When accurate and relevant, this information should help consumers to identify those products and services of the market with lower environmental impacts. In order to avoid impact shifts between different categories or life cycle stages, a life cycle approach should be applied when defining the rules for awarding eco-labels. Currently, only Type III eco-labels (or Environmental Product Declarations, EPDs) require that a LCA study of the product is undertaken following specific predefined calculation rules (named Product Category Rules, PCR). However, the technical and detailed contents of EPDs make them better suited for professional purchasers rather than final consumers, which may have the time and competence to understand their contents. On the other hand, Type I eco-labels are easier to understand, however the extent in which LCA methodology is followed in the definition of awarding criteria varies from one Type I scheme to another.

Within this context, the proposed model suggests that anchovies producers first develop an EPD of their products and then, by comparison to average market reference values of the different environmental impact categories (without aggregation), companies award an Eco-label Type I for their anchovies if they satisfy the threshold values. Such scheme, which may be applied to other product sectors, implies that the Eco-label Type I criteria should be based on LCA results of individual products. To this end, average environmental impacts of product categories should be known in advance. Therefore, LCA case studies will be developed for a number of anchovy's producers of the Spanish region of Cantabria in combination with scientific literature in order to define the thresholds values for the Type I eco-label.

Figure 3 shows conceptual scheme of Life Cycle Approach of the proposed model.

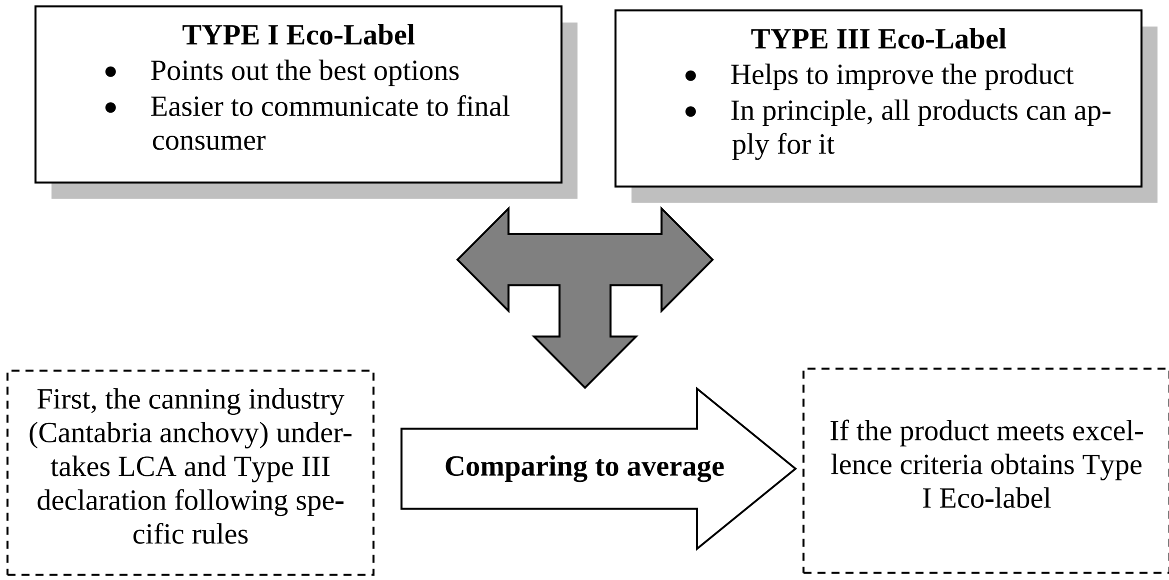

Figure 3. Conceptual scheme of Life Cycle Approach of the proposed model: Combining Type I and Type III Eco-Labels

3.1. Product Category Rules for development of Environmental Product Declarations of anchovy

The Type III environmental declarations, also known as "environmental product declarations" (EPDs), present relevant and quantitative environmental information about the life cycle of products. The information declared is based on an independently verified Life Cycle Assessment (LCA) study undertaken according to specific rules (i.e. Product Category Rules, PCR) developed in the framework of ISO 14040-44 and ISO 14025 standards. As stated in ISO 14025, the previous PCR document on fishery products will be taken into account, as well as previously published LCA studies on anchovies. In addition, we will be develop LCA studies of the products of local canning industries and the consultation with stakeholders of the Cantabria region that will allow gaining the required in-depth knowledge to develop a PCR document suitable for the Cantabria canned anchovies.

3.2. Definition of the environmental thresholds for the Type I eco-label

The Type I Eco-label requires a benchmark against which each applicant anchovy can be measured. For each impact category, average values can be defined based on the results declared through EPDs. These values should be updated periodically in order to foster the continuous reduction of the environ-mental footprint of anchovy.

Within the Project, pilot EPDs and LCA studies will be developed in collaboration with canning industries of the Cantabria Region. A comprehensive literature review of LCA studies of anchovies and related products in different parts of the world has been undertaken. Based on this available information, benchmarks will be identified for Cantabria canned anchovies for the following impact categories and indicators: global warming, water use and (fossil) primary energy consumption.

3.3. Expected Results

The development of the Product Category Rules and the definition of the environmental thresholds will be subject to external review by canning industries and other stakeholders belonging to the advisory group of the project. A number of seminars will be held with them to discuss key assumptions and methodological decisions, so as to achieve the widest possible agreement and contribute to the advancement of the state of the art in the LCA of canned anchovies.

Both the developed Product Category Rules and environmental thresholds for awarding the Type I eco-label will be tested for a reduced number of canning industries of the Cantabria Region. EPDs of Cantabria anchovies will be produced as pilot case studies within the Project. In addition, a PCR document will be available for its application in further cases.

With minor adaptations, the outcomes of the project could be transferred to other producing regions.

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Environmental Sustainability Assessment of an Innovative Process for partial Dealcoholization of Wines

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ABSTRACT

Global warming, viticulture progress and customer demand in aromatic wines have led to an international production of more and more alcoholized wines. More and more wine consumers complain about these high alcohol wines that are getting too heavy and strong to drink. Besides, this increase of alcohol content in wine is maybe not negligible from the viewpoints of individual alcohol intake and harmful effects of alcohol on health and behavior. The quest for techniques for wine partial dealcoholization allowing minimal aroma compound losses is yet currently ongoing. The currently most used techniques applicable at industrial scale are Reverse Osmosis and Spinning Cone Column. Both processes present high energy consumption and the sensory quality spoilage has not been overcome yet. Therefore, the main drawback of these technologies is the high energy consumption required, either to create vacuum conditions and slightly increase the working temperature or to pressurize the system. In order to reduce energy consumption, partial dealcoholization by Evaporative Pertraction (EP) technology is presented in this work as adequate technology to reduce energy consumption, and therefore, to reduce environmental impacts related to the dealcoholization process. In order to state the environmental benefits of obtaining the dealcoholized wines by the ecoinnovative process, the evaluation of the environmental impacts is necessary. Life Cycle Assessment (LCA) approach has been used in order to assess the environmental performance of the dealcoholized wines.

Keywords: Dealcoholization, Membrane contactors, Wine, Environmental Sustainability

1. Introduction

Wine is one of the most popular alcoholic drinks in the world. Mediterranean countries have a widespread culture of wine, being France, Italy and Spain the most important producers of this beverage in the world (Doering 2004).

Quality of wine is a key issue for the wine makers. Great effort is being done in optimizing the production of specific aromas and flavors (i.e. cherry, chocolate, vanilla), and minimize the formation of non-desired flavors (i.e. wet dog, plastic, rotten egg) (López et al. 2007). According to the European Commission (EC) regulations, wine is defined as an alcoholic beverage resulting from fermentation of grapes or grape must with ethanol content higher than 8.5% v/v (Commission Regulation, 2009). Generally, the wines are composed of 10–15% v/v alcohol, sugars, proteins, antioxidant agents and vitamins.

Alcoholic content has a strong impact on the quality of the wine affecting acidity, astringency and volatility of aroma compounds (Mermelstein 2000), altering the organoleptic properties of the product. The degree of ripeness of the grape conferring the optimum flavor characteristic matches normally the highest sugar content, and the resulting alcohol concentration. Therefore, a small adjustment in the alcohol content between 1 and 2% is currently and recently one of the most important objectives for the wine industry.

Nowadays, some methods to produce low alcohol-content wines or to adjust the ethanol content are employed by many wine makers in particular in the United States, for instance, spinning cone column (SCC) (Makarytchev 2004) and reverse osmosis (RO) (Ferrari 1991). Nevertheless, RO leads to a wine concentration (water and ethanol transfer through the RO membrane) which requires diluting further with water issued from wine itself. Though SCC is performed at mild operation temperatures (26–35 °C), this operation takes place in two steps: a first stage of aroma recovery and a second stage of ethanol removal. After ethanol separation, the aromatic fraction is added back to the wine, what results in a long and expensive operation.

Other technologies such as adsorption on zeolites and supercritical fluid extraction are being studied in the literature as possible alternatives to reduce the alcoholic content in beverages. Membrane technologies such as vacuum distillation, pervaporation and dialysis are also proposed to dealcoholize wine.

A membrane-based technology known as evaporative pertraction (EP), also named as osmotic distillation, shows promising results (Diban, el al. 2008) for partial dealcoholization of wine. During the EP process the feed phase (wine) is circulated through a hydrophobic hollow fiber membrane contactor while a second phase/stripping phase (water) flows through the other side of the membrane inside the hollow fiber contactor.

The partial pressure difference of the volatile components e.g. ethanol, between both phases creates the driving force of the process. The main advantages of the technology are: (i) the process can be conducted at room temperature, (ii) low energy consumption (no pressurization of the system is required) and (iii) a cheap and non-hazardous extractant, water, is normally used as stripping phase. The application of EP to get high dealcoholization degrees (> 2%, v/v) causes great sensory modifications on the wine. However, recently, the European Union (EU) regulation has fixed the maximum permitted dealcoholization level at 2% (v/v) (Commission Regulation 2009) for partially dealcoholized wines. Different red wine varieties partially dealcoholized (2%v/v) by EP were found to present an acceptable impact on the sensory properties (Diban et al. 2008). Moreover, the application of membrane contactors to the partial wine dealcoholization did not change significantly the presence of some of the main phenolic compounds and the color and total and volatile acidity of different red wine varieties studied (Gambuti et al. 2011).

In order to state the environmental benefits of obtaining the dealcoholized wine by the ecoinnovative process, the evaluation of the environmental impacts by the reference and by the alternative process is essential. Life Cycle Assessment (LCA) is a powerful tool used for assessing the environmental performance of a product, process, or activity that helps in identifying clean and sustainable alternatives in the process design activity. LCA also allows analysis at the different stages of the product life cycle. The present study focuses on the application of LCA for the evaluation of the dealcoholized wines by the different processes.

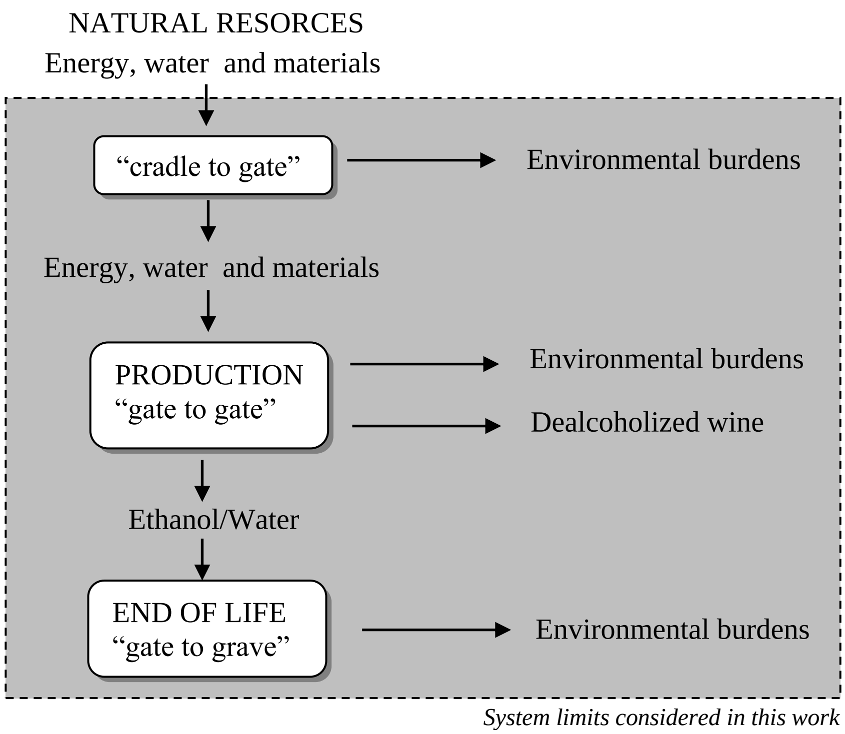


Figure 1. LCA for the environmental performance of dealcoholized wine using the reference and the alternative processes.

2. Methods

2.1. Goal and Scope

The main objective of the work is to quantify the environmental impacts of the conventional dealcoholization process (RO and SCC) and to evaluate the environmental benefits and drawbacks of EP process.

The scope of the assessment was based on the "cradle to grave" life cycle of a product and entailed resources usage and environmental impacts (Figure 1). The LCA started with the "cradle to gate" step where the natural resources water, energy, and materials needed for the manufacture of the resources used in the process were considered. The "gate to gate" step included traditional or the eco-innovative dealcoholization processes. The LCA ended with the "gate to grave" step that consisted of the transfer of the ethanol stream to valorization process or discharge.

2.2. Functional Unit

In this work, the functional unit (FU) was related to the dealcoholized wine, which is objective of the process under evaluation, the dealcoholization process. In order to compare the environmental performance of the traditional and eco-innovative process, the "cradle to grave" LCA of the different manufacture processes must be referred to the same quantity of the final product. The cubic meter of dealcoholized product was established as the most appropriate unit to describe the FU considering the available data. All the emission, consumption of materials, water, and energy during the scenarios are referred to this FU.

2.3. Description of Systems under Study.

In this work has been considered tree scenarios. Figure 1 illustrates the boundaries of the three scenarios under study.

Scenario 1: Evaporative pertraction (EP)

The evaporative pertraction (EP), also called osmotic distillation, is a membrane technology less energy demanding than Spinning Cone Column (SCC) and Reverse Osmosis (RO) as it operates at ambient temperature and atmospheric pressure. The gas transfer of volatile components (e.g. ethanol) is promoted from aqueous solutions through a micro-porous membrane. The wine is circulated through a hydrophobic hollow fiber membrane contactor while a second phase/stripping phase (water) flows through the other side of the membrane inside the hollow fiber contactor. The partial pressure difference of the components, between both phases creates the driving force of the process and permits the ethanol transfer. It has been determined that working under optimum operational conditions could minimize aroma compound losses below 20% during partial wine dealcoholization by EP. Therefore, the low energy consumption accompanied by the acceptable impact on sensory properties of wine make EP a promising technique to remove ethanol from wine at industrial scale (Diban et al., 2013, Lisanti et al., 2012 and Diban et al., 2008).

Scenario 2: Reverse osmosis (RO)

RO is the most used membrane separation process in wine industry for wine dealcoholization. However, the RO membranes allow the permeation of water and ethanol with high operation pressures (60 to 80 bar) which, in addition to considerable energy consumption, brings possible changes of the organoleptic properties of wine (Gonçalves et al., 2013). Two streams are obtained from the original wine: one of permeate containing water and ethanol, and one of retentate with the dealcoholized wine. The wine is slightly heated before the entrance to the membrane module from approximately 15ºC (wine storage temperature) to a temperature of 22-25ºC in order to facilitate the ethanol flux. The decrease in volume resulting from permeation is compensated by adding a large amount of water to the retentate. In this scenario, the water addition has been considered as an external income to simplify the calculations.

Scenario 3: Spinning Cone Column (SCC)

SCC is a two-stage distillation process used industrially for the production of wine with less 1% v/v of ethanol. In the first stage, the aroma compounds are removed at high vacuum conditions (0.04 atm), low temperature (26-28ºC) and collected in a high strength ethanol stream that represents approximately one percent of the original wine volume. The second stage in which ethanol is removed from the base wine is conducted at higher temperatures, usually around 38ºC. After ethanol reduction, the aroma fraction is added to the dealcoholized base wine. A number of ancillary devices are required for the SCC; heat exchangers to warm the product feed to operating temperatures, pumps and condensers to collect the gaseous vapor and collect the removed fraction. Therefore, this means a high capital outlay and operating costs (Belisario-Sánchez et al. 2009).

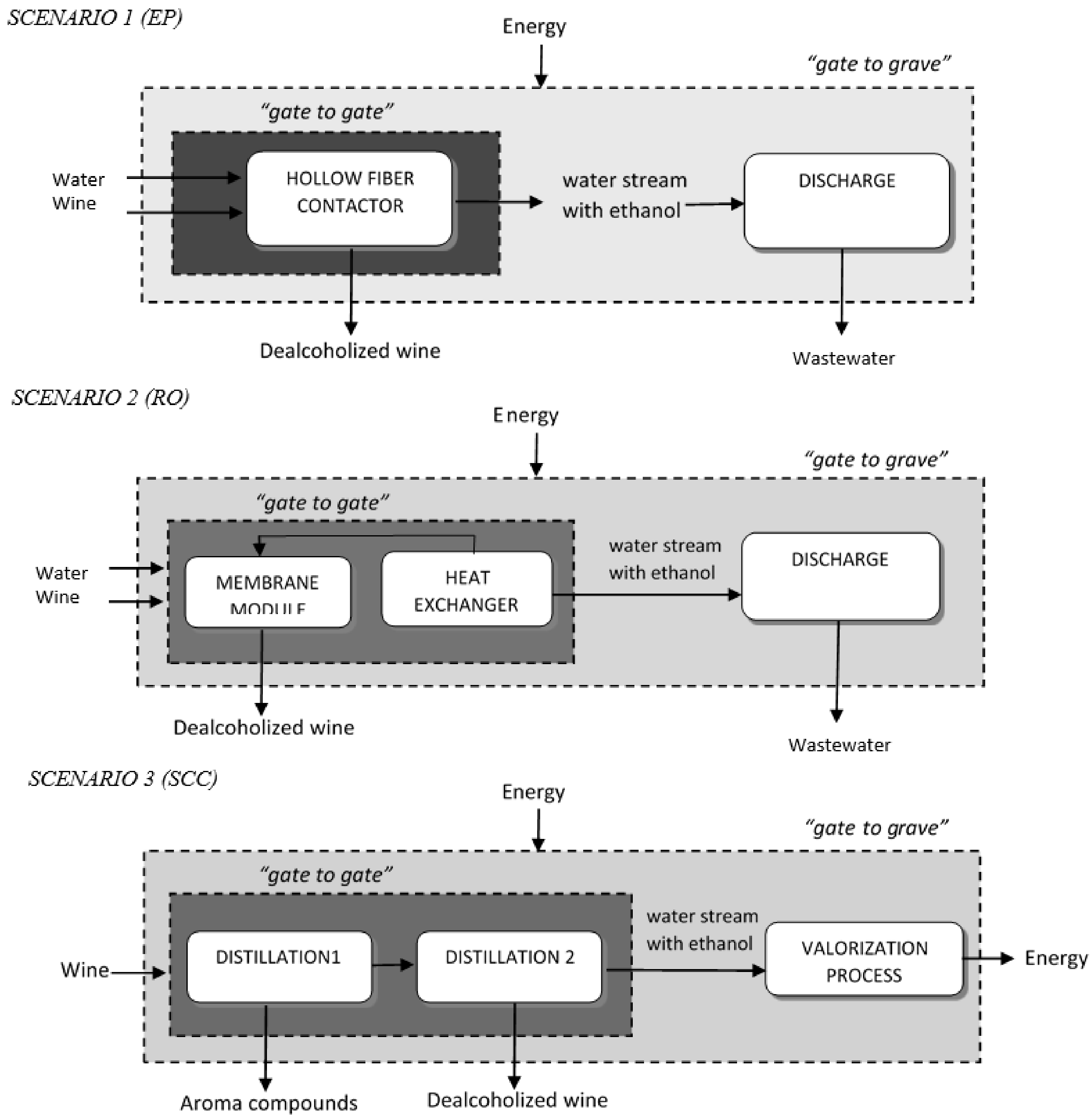


Figure 2. Flow diagrams of the "gate to gate" and "gate to grave" steps of the LCA: Scenario 1-Evaporative Pertraction (EP), Scenario 2-Reverse Osmosis (RO) and Scenario 3- Spinning Cone Column (SCC).

Table 1 collects a comparison of estimative energy and water consumption and the type of waste stream generated by the tree technologies (EP, RO and SCC). The waste generated in SCC is a stream rich in ethanol (<80%) that is usually valorized in waste to energy plants. The waste streams in RO and EP are poor amounts of ethanol (3-4%) that should be possible to recovery by distillation and pervaporation processes. However, these processes have low yields and high energy consumption, so the valorization process it is not technically and economically possible. The limits of BOD and COD established by the Spanish regulations for waste waters allow their discharge to the municipal sewage network.

2.4 Allocation

The total emissions and consumptions associated with dealcoholization process has been allocated to the wine stream. Additionally, waste incineration as valorisation process in the SCC process involves waste treatment and energy production, providing to the system an additional function. This situation was handled through system expansion. In this study the electric power mix of Spain included in the ELCD-PE GaBi database was selected as the technology in the system expansion (PE International 2011).

2.5. Life Cycle Inventory

The life cycle inventory (LCI) was developed using the data given by the dealcoholization unit supplier (AMTA, Alfa Laval), literature, regulation, ELCD-PE database (PE International. GaBi 4.4 Software and Databases for Life Cycle Assessment), chemical analysis, or was estimated by the authors using stoichiometric calculations. Further, Ecoinvent ELCD-PE database were mainly used for building the “cradle to gate” inventory. Table 1 encompasses the energy, water, and materials required in the scenarios, and Table 3 lists the generated outcome in the LCA steps.

Table 1. Comparison of estimative energy consumption, additional raw materials and waste generation between Scenario 1 (EP), scenario 2 (RO) and scenario 3 (SCC).

Dealcoholization process	**Scenario 1 (EP)**	**Scenario 2 (RO**	**SCC**
Energy consumption	Wine pump, water pump (<1 KWh/m3)	Wine pump, heat exchanger (approx. 1 KWh/m^3 for water desalination) (AMTA)	Vacuum pumps, heat exchangers, condensers (120 KWh/m^3) (Alfa Laval)
Raw materials	Water 0.5 m^3/m^3	Water 0.1 m^3/m^3 (Labanda et al., 2009)	-
Waste generation	Water stream with ethanol (<4% v/v) → discharge	Ethanol and water mixture (3.0-1.5% v/v) → discharge	Ethanol stream (80% v/v) → valorization

2.6. Life Cycle Impact Assessment

Most LCA studies apply the conventional impact assessment methods, such as CML 2001 (Guinée et al. 2001), EDIP 97 (Wenzel et al. 1997), or Eco-indicator 99 (Goedkoop et al. 2000). These methods use a set of metrics, which in some cases is hard to understand and makes difficult process comparison (Margallo 2014). In this sense, the use of novel indicators that reduce the LCA complexity and assist the decision making process, will improve the compression of LCA results. In this regard, this work propose a technical way to carry out the environmental sustainability assessment (ESA) of dealcoholization process based on a LCA approach using two main variables: natural resources sustainability (NRS) and environmental burdens sustainability (EBS) (Figure 3). NRS includes the consumption of the final useful resources, such as energy, materials, and water for the considered process and/or product. Land as a NR is currently excluded (Margallo et al. 2014). EBS is given by the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE). This set of indicators can be used to measure the environmental sustainability performance of an operating unit, providing a balanced view of the environmental impact of inputs (resource usage), and outputs (emissions, effluents, and waste) (IChemE 2002). In relation to the outputs, a set of environmental impacts to the atmosphere, aquatic media, and land was chosen. The environmental burden (EB) approach was used to estimate and quantify the potential environmental impacts (Garcia et al. 2013). In particular, the environmental impacts were classified in 12 variables grouped into the release to each environmental compartment: air, water, and land. These environmental impact categories chosen are a sub-set of those used internationally in environmental management, selected to focus on areas where the activities of process industry are most significant.

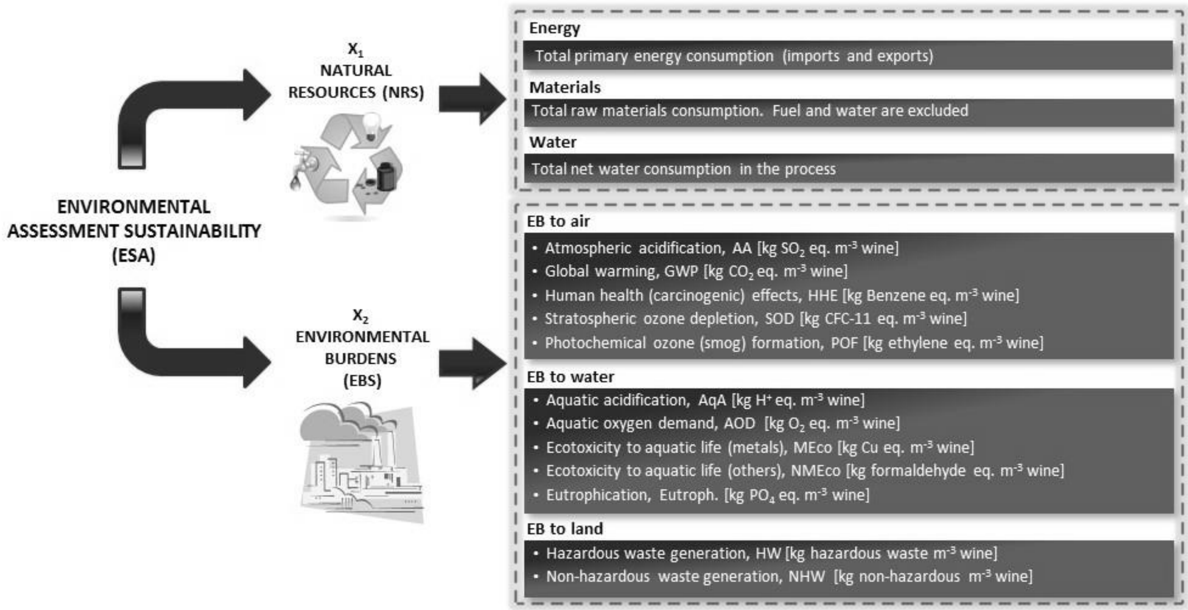


Figure 3. Life Cycle Impact Assessment methodology based on Natural Resources (NRS) and Environmental Burdens (EBS).

However, as natural resources (NR) and environmental burdens (EB) are rarely normalized, a normalization procedure is proposed. The normalization of EB is based on the threshold values of the European Pollutant Release and Transfer Register E-PRTR (E-PRTR Regulation 2006) leading to normalized variables, and a similar procedure based on the values given by Guide of Best Available Techniques of wine production (MTD Vi and Cava 2011) for the NR normalization. The E-PRTR regulation establishes the contaminants for which the European installations must provide notification to the authorities along with the threshold values of those pollutants. The threshold values can be used as an important aid in the normalization process because they provide an overview of the environmental performance of the installation at a European level (Margallo et al. 2014).

This normalization procedure reduces the complexity and allows the decision maker to track the progress towards environmental sustainability and to clarify the optimization procedure at least for the environmental pillar. As illustrated in Figure 1, the LCA considered the use of primary resources energy, water, and materials for obtaining the raw materials needed in the process or "gate to gate" cycle. This step generated some environmental burdens (EBs) caused by the substance upon the receiving environment. Further, the use of the resources needed in the process produced new EBs. The "gate to grave" step refers to the waste stream transfer to end of life process and also produced EBs, which refers to the discharge or energetic valorization. In this step, no materials as natural resources were considered. EBs for emissions to air and to water were estimated using GaBi 4.4. Related to the outputs, a set of environmental impacts to the atmosphere and aquatic media was chosen. The EBs approach was used to estimate and quantify the potential environmental impacts. The EB caused by the emission of a range of substance was calculated by adding the weighted emission of each substance. The weighting factor of the impact is known as the potency factor. In particular, the environmental impacts were classified into atmospheric and aquatic impacts. The EBs for emission to air were divided into atmospheric acidification (AA), global warming (GW), human health (carcinogenic) effects (HHE), stratospheric ozone depletion (SOD), and photochemical ozone (smog) formation (POF). The EBs for emission to water were defined by the aquatic oxygen demand (AOD), ecotoxicity to aquatic life (metals to seawater) (MEco), ecotoxicity to aquatic life (other substances) (NMEco), and eutrophication (Eutroph). The environmental sustainability indicators used in this study had different units depending on the environmental impact. In order to compare the EBs to air and water, the threshold values stated in the European regulation EC No 166/2006 for the main contributors to the environmental impacts were considered as weighting factors to obtain dimensionless impacts indicators. Table 2 shows threshold values from E-PRTR for Normalization and Impact Weighting purposes (Azapagic and Clift 1999, Regulation EC No 166/2006).

Table 2. Threshold Values from E-PRTR for Normalization and Impact Weighting purposes

Environmental Burden (EB)		Threshold Value (kg/year)	No. of substances
EB to air	AA (Kge SO_2)	150000	6
	GW (Kge CO_2)	100 million	23
	HHE (Kge benzene)	1000	52
	POF (Kge Ethylene)	1000	100
	SOD (Kge CFC-11)	1	60
EB to water	AOD (Kge H^+)	50000	14
	Meco (Kge Cu)	50	11
	NMEco (Kge formaldehyde)	50	18
	Eutroph (Kge phosphate)	5000	8

Abbrev: AA, atmospheric acidification; AOD, aquatic oxygen demand; EB, environmental burden; Eutroph, eutrophication; GW, global warming; HHE, human health effects; MEco, ecotoxicity to aquatic life (metals to seawater); NMEco, ecotoxicity to aquatic life (other substances); POF, photochemical ozone (smog) formation; SOD, stratospheric ozone depletion.

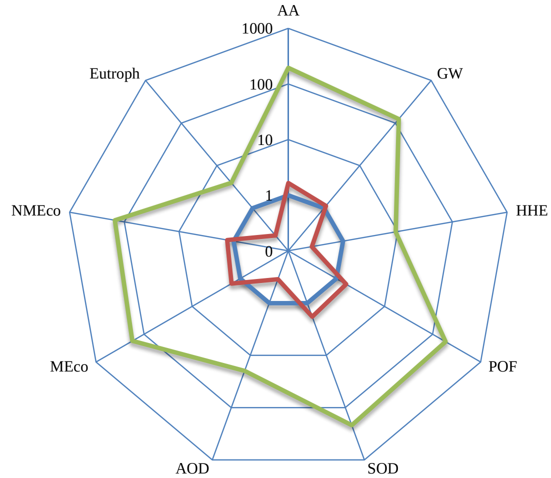
3. Results and discussion

The majority of the materials and energy used in scenario 1 (EP) are needed in the "cradle to gate" step to obtain water and primary energy: 100% of the materials and 64.7% of the energy as can be seem in Table 3. The energy used in the "gate to grave" step is neglected. Further, 78.0% of the water demand happens during the "cradle to gate" step. The "gate to gate" contributes to 20.0% of the water usage, mainly as second phase/stripping phase. It is important to note that the water footprint was out of the scope of this work, and only the use of natural resources has been considered when comparing the dealcoholization processes. Similar results has been obtained in scenario 2 (RO), where the natural resources to obtain water and primary energy (100% of the materials and 86.6% of the energy) has been mainly used in the life cycle or the process.

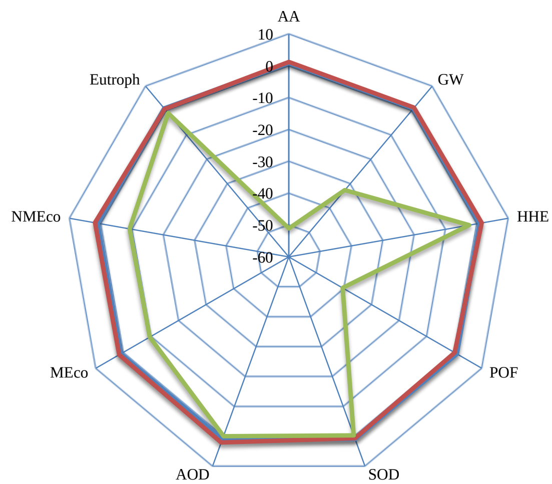
Table 3. Natural Resources Usage (NRS) in Scenario 1 (EP), Scenario 2 (RO), and Scenario 3 (SCC).

"cradle to gate"				
	units	**EP**	**RO**	**SCC**
Energy	(MJ)	7.76	12.30	1,433
Water	(kg)	1.95	1.85	195
Materials	(kg)	14.16	7.34	601
Basalt		1.90E-05	1.00E-05	8.32E-04
Bauxite		7.00E-03	1.41E-03	9.72E-04
Bentonite		7.31E-05	7.50E-05	8.06E-03
Clay		2.99E-05	4.33E-05	4.97E-03
Copper ore (0.14%)		4.56E-04	2.58E-04	2.22E-02
Gypsum (natural gypsum)		1.33E-05	2.21E-05	2.60E-03
Heavy spar (BaSO4)		1.80E-04	1.81E-04	1.93E-02
Inert rock		1.06E+00	1.76E+00	2.06E+02
Iron ore (56-86%)		3.00E-03	7.34E-04	1.78E-02
Lead - zinc ore (4.6%-0.6%)		4.38E-04	1.00E-04	1.68E-03
Limestone (calcium carbonate)		4.21E-02	1.21E-02	4.85E-01
Magnesium chloride leach (40%)		4.88E-03	1.14E-03	2.23E-02
Natural Aggregate		7.49E-03	4.98E-03	4.64E-01
Quartz sand (silica sand; silicon dioxide)		1.28E-01	2.55E-02	2.69E-03
Sodium chloride (rock salt)		6.51E-03	1.31E-03	1.09E-03
Soil		4.27E-02	9.24E-03	9.34E-02
Zinc - copper ore (4.07%-2.59%)		2.14E-04	5.53E-05	1.67E-03
Zinc - lead - copper ore (12%-3%-2%)		4.96E-05	1.73E-05	9.84E-04
Air		1.29E+01	5.52E+00	3.93E+02
Carbon dioxide		1.54E-02	3.02E-02	3.61E+00
"gate to gate"				
	units	**EP**	**RO**	**SCC**
Energy	(MJ)	3.60	1.80	432
Water	(kg)	0.5	0.1	-
Materials	(kg)	-	-	-
"gate to grave"				
	units	**EP**	**RO**	**SCC**
Energy	(MJ)	0.40	0.11	-2,394.5
Water	(kg)	0.15	0.03	-
Materials	(kg)	-	-	-

As can be seem in Table 3, energy in the “gate to grave” step of the scenario 3 (SCC) is negative related to the valorization process of the waste stream to energy. It is possible to check that scenario 3 (SCC) has a greater impact on energy consumption in the gate to gate step than RO and EP, but lacks of the necessity of any additional raw material, while RO and EP needs to supply water to perform the alcohol adjustment. Energy consumption appears as a key factor in the wine dealcoholization process. In this sense, energetic impact is environmental friendly when energetic valorization of the waste stream is possible.



(a) “cradle to gate” and “gate to gate”



(b) “cradle to gate”, “gate to gate” and “gate to grave”

Scenario 1 (EP) Scenario 2 (RO) Scenario 3 (SCC)

Figure 4. Weighted and normalized environmental impacts (EBS) of Scenario 1 (EP), Scenario 2 (RO), and Scenario 3 (SCC).

The weighted and normalized environmental impacts referred to the E-PRTR threshold are shown in Figure 4. Figure 4 (a) shows the environmental impact considering “cradle to gate” and “gate to gate” steps. From figure 4 (a) it is possible to check that the total environmental impact of scenario 1 (EP) is similar to scenario 2 (RO). However, the environmental impact of scenario 3 (SCC) increased significantly. The environmental impact to water and to air were mainly based on the contribution of the energy consumption of SCC technology.

From Figure 4 (b) it is possible to note that, when the valorization process of the water/ethanol stream is considered, environmental impacts of the scenario 3 (SCC) are drastically reduced. However, the actual energy valorization of ethanol should account on an additional evaporation of water that has not been considered in the present scenario. This should be further analyzed. According to this, AA, GW and POF impacts decrease about 130%. The environmental impacts of the scenario 3 (SCC) are negative, which is related to the avoided burdens on the generation of energy in the valorization process of the waste stream.

5. Conclusion

The LCA assessment of the dealcoholization practices demonstrated that the environmental profile of the “cradle to gate”, “gate to gate”, and “gate to grave” steps are directly related and that the “cradle to gate” and “gate to grave” (when valorization process in considered) steps of the scenarios contributed significantly more to the environmental impacts than the “gate to gate” step. The production of primary energy has the most important contribution to the environmental impact of the scenarios.

In this work, the reduction of the environmental impact of the partial dealcoholization of wines was obtained by reducing energy consumption. However, other measures may be implemented to the “cradle to gate” and “gate to grave” in order to further reduce the environmental impact of the overall LCA of the EP process. These measures may consist of some valorization process, avoiding the discharge of the waste stream. This work concludes that the eco-innovative dealcoholization EP process is positive in terms of resource usage and EB.

Finally, this work shows that future research should focus on evaluating the economic and social costs related to the eco-innovative dealcoholization process, in order to assess the sustainability of the process.

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Influence of site conditions and production system on the environmental impacts of domestic and imported cheese

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ABSTRACT

In this study, cheese produced in Switzerland was compared to imported cheese from the main countries of origin, France, Italy, and Germany. The environmental impacts of cheese production were dominated by milk production. The average Swiss milk production was characterized by a low input of concentrates and the lowest milk yield per cow. Despite this lower milk yield, the environmental impacts of Swiss milk were lower or similar to milk produced in the other countries, with the exception of land occupation. The good growing conditions for grassland and a high quality of roughage allow to produce milk at moderate intensity level in an efficient way. The study showed that there is no simple relationship between the amount of concentrates, milk yield and environmental impacts, instead the results depend on the whole production system, and site conditions can influence the environmental impacts significantly.

Keywords: milk, cheese, imports, concentrate feed, milk yield

1. Introduction

1.1. Background

Milk is the most important product of Swiss agriculture and also dominates its environmental impacts. In 2012, 21 % of the total agricultural production value in Switzerland was generated with milk (BLW 2013). Only 12 % of Swiss milk was directly consumed, whereas 42 % was transformed to cheese (BLW 2013). Since 2007, a free-trade agreement is effective between Switzerland and the EU, leading to increased imports and exports of cheese in Switzerland. In an effort to ensure the future ability of Swiss agricultural products to compete with those from abroad, the food industry, supported by the Federal Government, has developed a quality strategy aimed at setting the environmental and quality credentials of Swiss farm products apart from those of other countries. There is, however, a shortage of data which would allow a systematic and scientifically sound comparison of the environmental impact of foodstuffs from different countries of origin. Furthermore, production conditions as well as the production systems within a country vary widely. For a sound decision making on how to improve the environmental impacts of Swiss products, there is a need for reliable and comparable data on the environmental impacts of agricultural products originating from different countries.

1.2. Goal and scope definition

The aim of this study was to generate inventories of cheese produced in Switzerland and abroad, and to compare the environmental impacts of Swiss and imported cheese. Cheese produced in Switzerland was compared to cheese from the main countries of origin, France, Italy, and Germany. The Swiss production was differentiated according to the region (lowlands, hills, and mountains) and to the production system (grassland based with barn feeding, full grazing system and high-yielding system with higher concentrate input). To be able to distinguish between differences in the agricultural production and differences occurring in downstream stages (e.g. different transport distances), life-cycle-analyses were conducted both at the farm gate and at the point of sale. For the comparison at the farm gate and at the point of sale the functional units were 1 kg milk and 1 kg of cheese, respectively.

2. Methods

The environmental impact of the products investigated was determined using SALCA (Swiss Agricultural Life Cycle Assessment; Nemecek et al. 2010), the life cycle assessment method developed by Agroscope. SALCA comprises a life cycle inventory database for agriculture, models for direct field and farm emissions, a choice of methods for impact assessment, calculation tools for farming systems (farm and crop level), an evaluation concept, and a communication concept for the results.

The following environmental impacts were examined: non-renewable energy demand, global warming potential, ozone formation potential, demand for phosphorus and potassium resources, land competition, deforestation, water use WSI (water use in m^3 weighted with a "water stress index" which takes account of water scarcity in the different countries), eutrophication potential, acidification potential, terrestrial ecotoxicity potential, aquatic ecotoxicity potential and human toxicity potential. The impact assessment methods used as well as the models used to calculate direct emissions are described in Nemecek et al. (2010). A rating system was used to assess the differences in individual results. It was not possible to examine the environmental impact of soil quality or biodiversity due to a lack of the required data. The differences between the impacts were evaluated by assessment classes, which differed among the impact categories.

2.1. System boundaries

For the agricultural production, all on-farm activities and external inputs (e.g. feedstuff, diesel, mineral fertilizers) for milk production were considered, as well as the necessary infrastructure (buildings and machinery) and the usable agricultural area needed (Figure 1, A).

For cheese production, additionally to milk production transports to the cheese dairy, processing and transports to the point of sale in Switzerland were considered (Figure 1, B). Cheese processing generally took place in the country of origin and the finished cheese was transported to the point of sale in Switzerland.

2.2. Production inventories

The Swiss milk production systems were derived from model farms of a former project (Hersener et al., 2011) and refer to an average Swiss production. The foreign inventories were newly created on the basis of the Swiss systems, the most important key figures for livestock production being adapted to the specific country. If no data were available, Swiss data were extrapolated on the basis of the number of dairy cows. Table 1 shows an overview of the most important production inventories of the specific countries. The Swiss data represent a national average; the foreign systems refer to the most important production systems (typical) within the country under consideration.

An economic allocation was conducted between milk and meat. 87 % of the inputs were allocated to milk production and 13 % to meat production. In order to ensure comparability, Swiss data were used also for the foreign systems. Background data were derived from the SALCA database (Nemecek et al., 2010) and ecoinvent V2.2 (ecoinvent Centre, 2010).

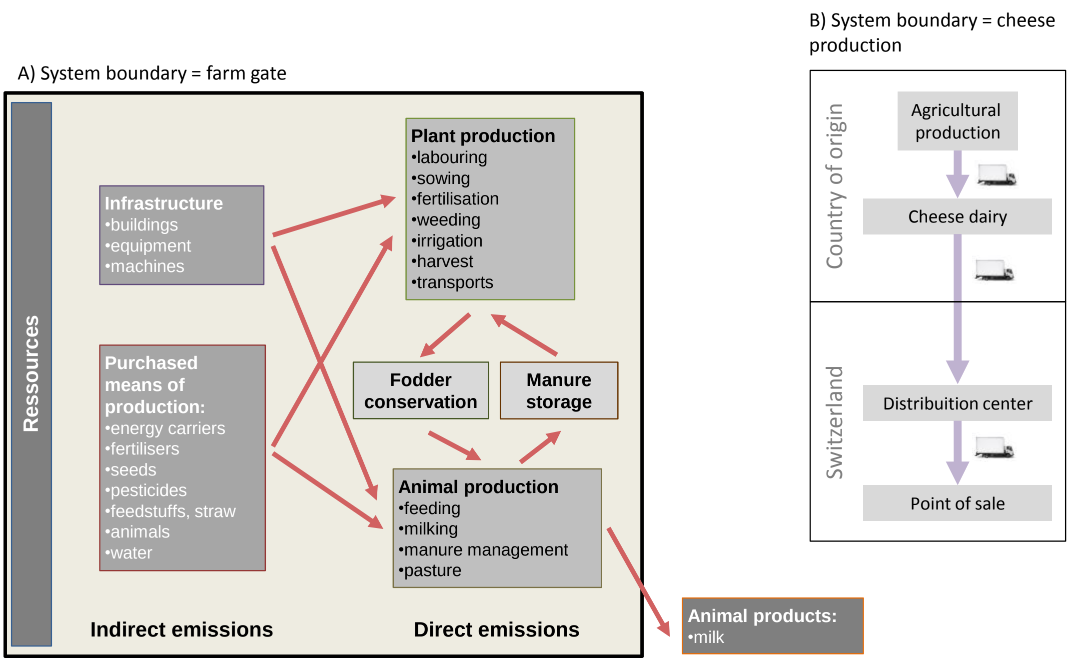

Figure 1. System boundaries for agricultural production (A) and cheese production (B)

Table 1. Production parameters of the milk production systems in Switzerland (Milk CH), Germany (Typ. Milk DE), France (Typ. Milk FR) and Italy (Typ. Milk IT). DM: dry matter, FM: forage mixture, BR: basic ration. Typ. = typical.

Parameter	***Unit***	**Milk CH**	**Typ. Milk DE**	**Typ. Milk FR**	**Typ. Milk IT**
Number of dairy cows		19.9	80	46	419
Age of first calving	*months*		28	29	27
Useful life	*months*	40	37.7	32.9	21
Restocking	*%*	30	36	37	37
Calves born alive	*calves*year^{-1}*	0.9	0.9	0.83	0.9
Barn arrangement		50% free stall barn	50% free stall barn	100% free stall barn	100% free stall barn
Milk yield	*kg*cow^{-1}*year^{-1}*	6,800	8,000	8,200	9,450
Milk production	*kg*farm^{-1}*year^{-1}*	127,372	600,000	369,000	3,721,026
UAA for milk production	*ha*	0.58	0.6	0.84	0.51
Pasturing	*days*year^{-1}*	167	-	112	-
Feed intake	*kg DM*cow^{-1}*day^{-1}*	20.2	19.6	20.4	18.0
Concentrates	*kg DM*LU^{-1}*year^{-1}*	877 *25% FM dairy* *75% FM cereal*	2,019 *33% wheat* *33% barley* *25% soya* *8% rape meal*	2,164 *47% soy meal* *41% wheat* *8% FM conc.* *4% FM min.*	2,498 *35% maize flour* *23% soy meal* *15% cotton seed* *13% protein supp.* *10% soy seeds* *4% maize flakes*
Basic ration	*kg DM*LU^{-1}*year^{-1}*	6,752 *41% grass silage* *29% grass* *19% hay* *11% maize silage*	5,100 *56% grass silage* *37% maize silage* *12% hay*	5,804 *62% maize silage* *26% grass* *9% grass silage* *3% hay*	4,068 *60% maize silage* *39% hay* *1 % grass*
Share of basic ration in total ration	%	89	72	76	62

2.3. Downstream processes

For all systems analyzed, data about cheese production were derived from Schmid et al. (2010). The cheese dairy in this study represents the average cheese production in artisanal dairies in Switzerland. Due to the lack of more country-specific data, this data was also used for the foreign systems; only the electricity mix was adapted to the country in question. An economic allocation between cheese and its by-products (whey) was carried out at which 85 % of the environmental impacts were allocated to cheese.

For the transports, average transport distances from the most important milk producing regions of each country (after Hemme et al., 2011) were considered. Table 2 shows an overview of the means of transport and transport distances assumed.

Table 2. Means of transport and transport distances for Swiss and imported cheese

Country of origin, route of transport	Means of transport	Distance (km)
Switzerland		
Farm – cheese dairy (milk)	lorry 3.5-20 t, fleet average Switzerland, refrigerated	20 [1)]
Cheese dairy – distribution center (cheese)	lorry 20-28 t, fleet average Switzerland, refrigerated	100 [2)]
Distribution center – point of sale (cheese)	lorry 3.5-20 t, fleet average Switzerland, refrigerated	25 [2)]
Germany		
Farm – cheese dairy (milk)	lorry >16 t, fleet average Europe, refrigerated	150 [3)]
Cheese dairy – distribution center (cheese)	lorry >16 t, fleet average Europe, refrigerated	650 [4)]
Distribution center – point of sale (cheese)	lorry 3.5-20 t, fleet average Switzerland, refrigerated	25 [2)]
France		
Farm – cheese dairy (milk)	lorry >16 t, fleet average Europe, refrigerated	150 [5)]
Cheese dairy – distribution center (cheese)	lorry >16 t, fleet average Europe, refrigerated	800 [4)]
Distribution center – point of sale (cheese)	lorry 3.5-20 t, fleet average Switzerland, refrigerated	25 [2)]
Italy		
Farm – cheese dairy (milk)	lorry >16 t, fleet average Europe, refrigerated	150 [5)]
Cheese dairy – distribution center (cheese)	lorry >16 t, fleet average Europe, refrigerated	400 [4)]
Distribution center – point of sale (cheese)	lorry 3.5-20 t, fleet average Switzerland, refrigerated	25 [2)]
Sources: [1)]Schmid (2010); [2)]Alig et al. (2012) [3)]Reinhard et al. (2009); [4)]Own estimation (Google Maps), based on main production regions according to Hemme (2011); [5)]Assumption: as in Germany		

3. Results

3.1. Agricultural production

At farm gate level Swiss milk production generally scored more favorably or was within the same range as milk production abroad. The only exception was land competition, where the Italian system was lower. Table 3 gives an overview of the environmental impacts of the different milk production systems analyzed.

The higher use of concentrates in the foreign systems led to a higher demand in phosphorus and potassium resources and to considerable higher values for deforestation, which was caused by the higher use of soya in the feed ration. Additionally, the foreign systems had a higher water use, which was a direct consequence of the higher irrigation during the cultivation of the concentrates used. Only for land competition one foreign system (Italy) achieved lower values than the Swiss system. This was due to its concentrate based ration with a very low share of grass, which leads to a high energy yield per hectare land occupied. For all other environmental impacts, the higher milk yield through increased use of concentrates in the foreign systems did not result in lower environmental impacts. On the contrary, the energy required to produce one kilogram of milk increased with the milk yield per cow due to the purchase of extra feed and the use of energy carriers on the farm, both of which were higher in foreign systems than in Switzerland (Figure 2). In addition, the higher restocking rates in the foreign systems lead to a higher impact of the purchased animals.

Regarding the global warming potential, the higher milk yields in the foreign systems led to lower methane emissions; however, for the total global warming potential there was no difference between the systems analyzed. The lower methane emissions in the foreign systems were compensated by higher CO_2-emissions caused by the higher energy demand (Figure 3). The same mechanism was found for ozone formation, where the lower methane emissions were compensated by higher nitrous oxide emissions.

Table 3. Overview of the environmental impacts per kg milk of milk production in Switzerland (Milk CH), Germany (Typ. Milk DE), France (Typ. Milk FR) and Italy (Typ. Milk IT). WSI: water stress index, N: nitrogen, P: phosphorus

Environmental impact	***Unit***	**Milk CH**	**Typ. Milk FR**	**Typ. Milk DE**	**Typ. Milk IT**
Non-renewable energy demand	*MJ-Eq.*	4.31E+00	4.64E+00	4.84E+00	6.41E+00
Global warming potential	*kg CO_2-Eq.*	1.26E+00	1.32E+00	1.31E+00	1.21E+00
Ozone formation potential (vegeta-tion)	*m^2*ppm*h*	1.38E+01	1.36E+01	1.44E+01	1.33E+01
Ozone formation potential (human)	*Person*ppm*h*	1.08E-03	1.06E-03	1.12E-03	9.96E-04
demand for potassium resources	*kg*	9.63E-04	6.57E-03	2.63E-03	7.47E-03
Demand for phosphorus resources	*kg*	1.05E-03	1.97E-03	2.15E-03	1.44E-03
Land competition	*m^2a*	1.71E+00	1.57E+00	1.75E+00	1.42E+00
Deforestation	*m^2*	4.30E-04	7.28E-03	1.08E-02	1.77E-02
Water use (WSI)	*m^3*	9.00E-04	1.61E-03	2.47E-03	5.49E-03
Terrestrial Eutrophication potential	*m^2*	9.12E-01	1.10E+00	9.96E-01	8.27E-01
Aquatic eutrophication potential N	*kg N*	4.64E-03	6.57E-03	5.52E-03	4.37E-03
Aquatic eutrophication potential P	*kg P*	1.85E-04	3.60E-04	3.11E-04	3.28E-04
Acidification potential	*m^2*	2.24E-01	2.69E-01	2.46E-01	2.14E-01
Terrestrial ecotoxicity potential	*kg 1,4-DB-Eq.*	5.99E-04	1.20E-03	6.88E-04	7.61E-04
aquatic ecotoxicity potential	*kg 1,4-DB-Eq.*	9.08E-02	1.11E-01	7.87E-02	1.10E-01
Human toxicity potential	*kg 1,4-DB-Eq.*	2.12E-01	2.18E-01	2.12E-01	2.67E-01

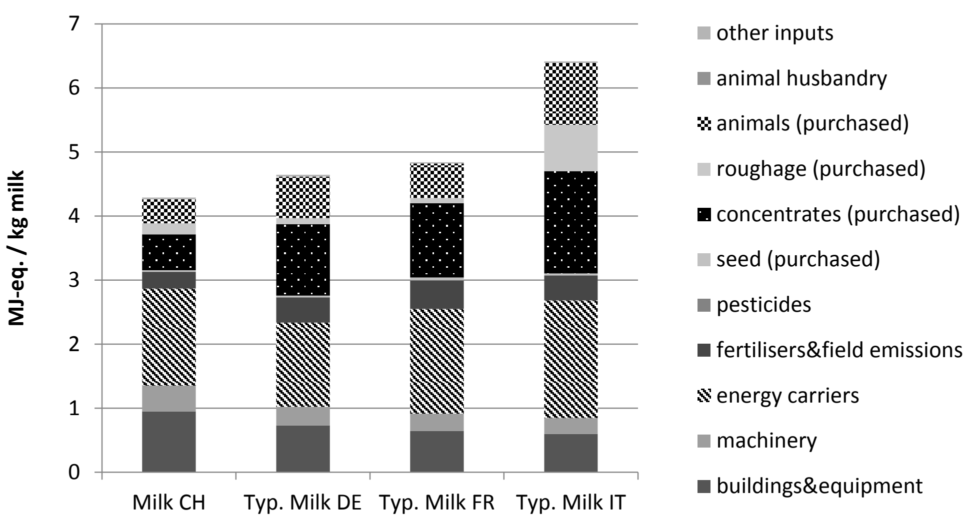

Figure 2. Energy demand of the different milk production systems analyzed.

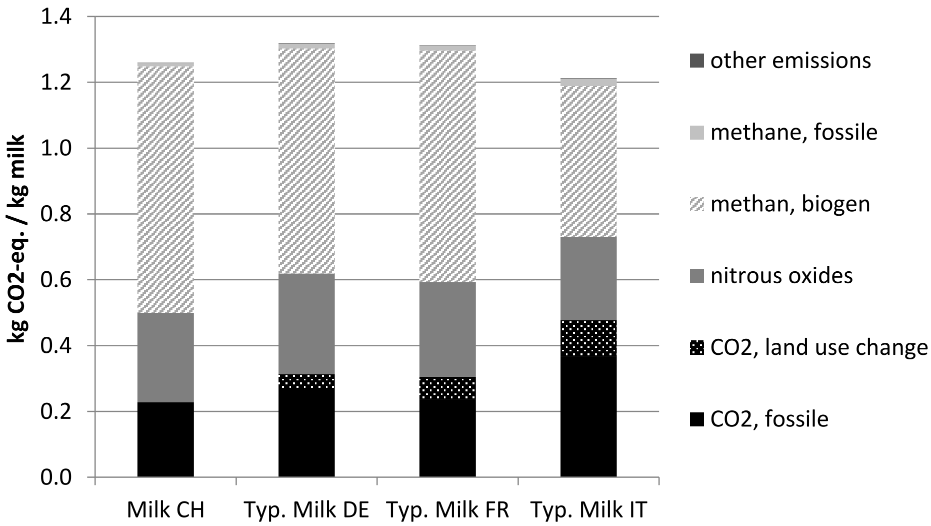

Figure 3. global warming potential of the different milk production systems analyzed

For the nutrient related impacts, the foreign systems had a higher aquatic eutrophication potential from phosphorus compounds. This was mainly due to higher field emissions during cultivation of the feedstuff used. The German system was also higher for the other nutrient related impacts (terrestrial eutrophication potential, aquatic eutrophication potential phosphorus and acidification potential). In Germany, the fertilization level on the agricultural area used was highest; in particular there was the highest amount of slurry applied per hectare, which lead to high emissions of ammonia.

For the toxicity related impacts, there were no relevant differences between the systems analyzed. This was also due to the high uncertainties in this field, so that relatively big differences are still not considered as being relevant. Only for the terrestrial ecotoxicity potential the German system had higher impacts than the Swiss system.

Comparing dairy systems within Switzerland showed that mountain production had generally higher impacts than lowland production due to less favorable conditions like lower yields, longer winter feeding periods and steep slopes in the mountains. Grassland based dairy systems with moderate milk yield had lower ecotoxicity impacts, lower use of mineral resources (P and K) and lower contribution to deforestation than the standard dairy system, but similar or higher impacts regarding the other categories.

3.2. Results at point of sale

The environmental impacts per kilogram of cheese followed the same pattern as the environmental impacts per kilogram of milk, since the agricultural phase dominated the environmental impact of cheese production up to the point of sale. The downstream process with the strongest influence was the cheese dairy, followed by the transports. The impact of the distribution center was negligible.

4. Discussion

The good growing conditions for grassland with abundant precipitation and a high quality of roughage in Switzerland allow to produce milk at moderate intensity levels with small amounts of concentrates in an efficient way. A higher use of concentrates with the resulting higher milk yield in the foreign systems did not lead to an improvement of the environmental performance of those systems. On the other hand, the analyses of modeled milk production systems with different share of grass in the feed ration within Switzerland showed a lower performance for the systems with a high share of grass in the diet.

This shows that when comparing different feeding strategies, not only the directly affected parameters such as feed intake and type of feed should be considered, but all relevant production parameters as well as site conditions must be included. The impact of a feeding strategy on health parameters and service life of dairy cows are particularly important. The foreign systems consistently exhibited a lower service life and a higher replacement rate than the Swiss systems, which significantly contributed to the higher energy requirement.

For the methane related impacts, this study revealed no difference between the Swiss and the foreign systems. However, methane emissions strongly depend on the equations used to calculate the emissions from enteric fermentation, as shown by Hagemann et al. (2011). According to the method of the Intergovernmental Panel on Climate Change (Eggleston et al. 2006) used in this study the methane emissions increase proportionally to the gross energy intake of dairy cows. The statistical analysis of a large number of published experimental data carried out by Ramin & Huhtanen (2013) confirms that gross energy intake is the main driving factor of methane emissions. However, the quality of the estimate could be improved by including several characteristics of the feed ration. Further studies are needed to estimate the methane emissions of grass based systems compared to concentrate based systems.

Two other essential parameters for the environmental effects of milk production are the manure management and manure spreading. While a high proportion of manure in the total fertilization has a positive impact on energy demand as well as on resource use phosphorus and potassium, there is a negative influence on the nutrient-related environmental impacts due to the slurry-related ammonia emissions. The modeling of the individual milk production systems revealed to be particularly difficult in this regard, as for the individual countries only few data on manure spreading on crop level was available and there was a complete lack of information about application dates. The management of the on-farm agricultural area for the production of roughage in the foreign systems is therefore partially modeled with Swiss data. Including country specific data on manure management and manure spreading could change the results in particular for the nutrient related impacts.

Another difficulty is the high variability in the milk production systems within a country. The Swiss results showed that the environmental impacts of different milk producing systems within a country can vary significantly. Guerci et al. (2013) compared twelve different milk producing farms in Denmark, Germany and Italy and analyzed their energy demand, global warming-, acidification- and eutrophication potential per kilogram of milk produced. He stated a significant positive effect of a high proportion of grassland on energy use, global warming potential and acidification. However, as only 12 farms were assessed, a more general analysis of the relation between the production and the environmental impact has to be handled with care, especially as the study revealed huge variability in environmental impact within the group of farms analyzed (Guerci et al. 2013).

To compare the milk production of a country with the one of another country, all milk production systems existing in a country would need to be modeled to form the respective national average. The milk producing systems in this study refer to a typical system and – except in the Swiss case – not to the average milk production of the country concerned. This must be considered when interpreting the results. Nevertheless, this study shows which parameters are important for the environmental impact of milk production and provides important information on existing differences between the countries studied.

5. Conclusion

The environmental impacts of cheese production are dominated by milk production. The study showed that there is no simple relationship between amount of concentrates, milk yield and environmental impacts; instead the results depend on the whole production system. Site conditions can influence the environmental impacts significantly: In Switzerland, the good growing conditions for grassland with abundant precipitation and a high quality of roughage allow to produce milk at moderate intensity level with small amounts of concentrates in an efficient way.

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Estimating the water footprint of milk produced in the southern region of Brazil

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ABSTRACT
Water use is a hot topic worldwide in sustainability assessment. However, this issue is not well evaluated in traditional life cycle assessment (LCA). On the other hand, water footprint (WF) is a methodology that has been developed to give a more complete overview in the water use of particular products (Hoekstra et al. 2011). In this sense, this study evaluated the blue and green WF of three different milk production systems in the southern region of Brazil. To calculate the WF we based on primary data and several data from literature. The results showed that milk from confined feedlot, semi-confined feedlot, and pasture-based systems had blue WF of 19, 11, and 7 liters/kgECM, respectively, and green WF of 1478, 2209, and 1584 liters/kgECM, respectively. We could conclude that higher pasture productivities and/or feed conversion ratio should be sought in all systems, in order to reduce the green (and overall) WF.

Keywords: milk, water footprint, LCA, Brazil

1. Introduction

Water is a renewable resource, although its availability in good environmental quality is an important issue worldwide, especially in dry areas. In this sense, water use becomes an important issue for environmental sustainability. There are several ways to evaluate the environmental sustainability of products, and the most predominant is Life Cycle Assessment (LCA) (Dewulf and Van Langenhove 2006). Even though some efforts have been made in recent years regarding water use impacts in LCA (Bayart et al. 2010; Kounina et al. 2013; Pfister et al. 2009), in most traditional LCA studies it is not well evaluated (Milà i Canals et al. 2009), especially when dealing with food products.

Another environmental sustainability methodology, called water footprint (WF), is able to deal with water use in a more complete way. WF is a methodology that has been developed to give a more complete overview in the water use of particular products (Hoekstra et al. 2011). The combination of LCA and WF for sustainability assessment has already been discussed in literature (Boulay et al. 2013; Jefferies et al. 2012; Milà i Canals et al. 2010; Milà i Canals et al. 2009).

Milk is a product from the agricultural sector, which has (as any other product) a certain environmental footprint. In this sense, its environmental impacts have been studied in many reports and scientific publications, although not much has been done for milk in Brazil.

A LCA of milk produced in the southern region of Brazil has been recently studied (Léis 2013), but it did not consider water use and water use impacts. Therefore, the objective of this study is to account for the blue and green WF of three different production systems in the southern region of Brazil. In this sense, it can complement the aforementioned study, providing an additional report for a more complete environmental profile of that product.

2. Methods

This study evaluated three different milk production systems in the southern region of Brazil (Parana and Santa Catarina states).

The first system is a confined feedlot system, and it is located in the city of Mandaguari, north of Parana state. In this system the cows solely receive animal feed in the trough. The feed is composed by cottonseed, silage, commercial feed concentrate (cottonseeds, maize grains, wheat bran, soybean hulls, premix), hay, minerals, premix and other cattle foodstuffs.

The second system is a semi-confined feedlot system where the cows are fed in the trough and also through grazing. Apart from the grazed material, the feed is composed by silage, citrus pulp (byproduct of the orange juice industry), brewers spent grain (byproduct of the beer industry), commercial feed concentrate (cottonseeds, maize grains, wheat bran, soybean hulls, premix), minerals, premix, forage and other cattle foodstuffs. This system is located in the city of Porto Amazonas, east of Parana state.

In the third system the cows are mainly fed through grazing, but some feed is still provided in trough. This feed is composed by maize, soybean meal and mineral salts. This system is located in Campos Novos, central region of Santa Catarina state.

The functional unit was 1 kg of energy corrected milk (ECM) at the farm gate. The life cycle inventory was based on Léis (2013), in which a LCA was performed with focus on other environmental impact categories, as eutrophication and carbon footprint.

We calculated the WF based on Hoekstra et al. (2011). For blue water we considered estimations of animal water consumption, based on Araújo et al. (2011), which elaborated an equation of water consumption based on dry matter consumption, milk production, salt consumption, and minimum daily average temperature. We also accounted for the consumption of water from cleaning processes at dairy farms, based on Guerra et al. (2011), considered to be $25L/m^2$. With that, and knowing the area used for milk production in all three systems, we were able to calculate the direct use of blue water. For the indirect consumption of blue water, i.e., the blue water consumed to produce the feed ingredients, for instance, we used data based on literature (mainly the ecoinvent (2010)).

None of the ingredients consumed by the cows were considered to be produced in irrigated systems. According to IBGE (2006), in Brazil only 8% of cotton, 6% of maize, 4% of soybean, and 7% of barley are produced in irrigated systems. Therefore, due to representativeness, we assumed that those products were produced in non-irrigated systems.

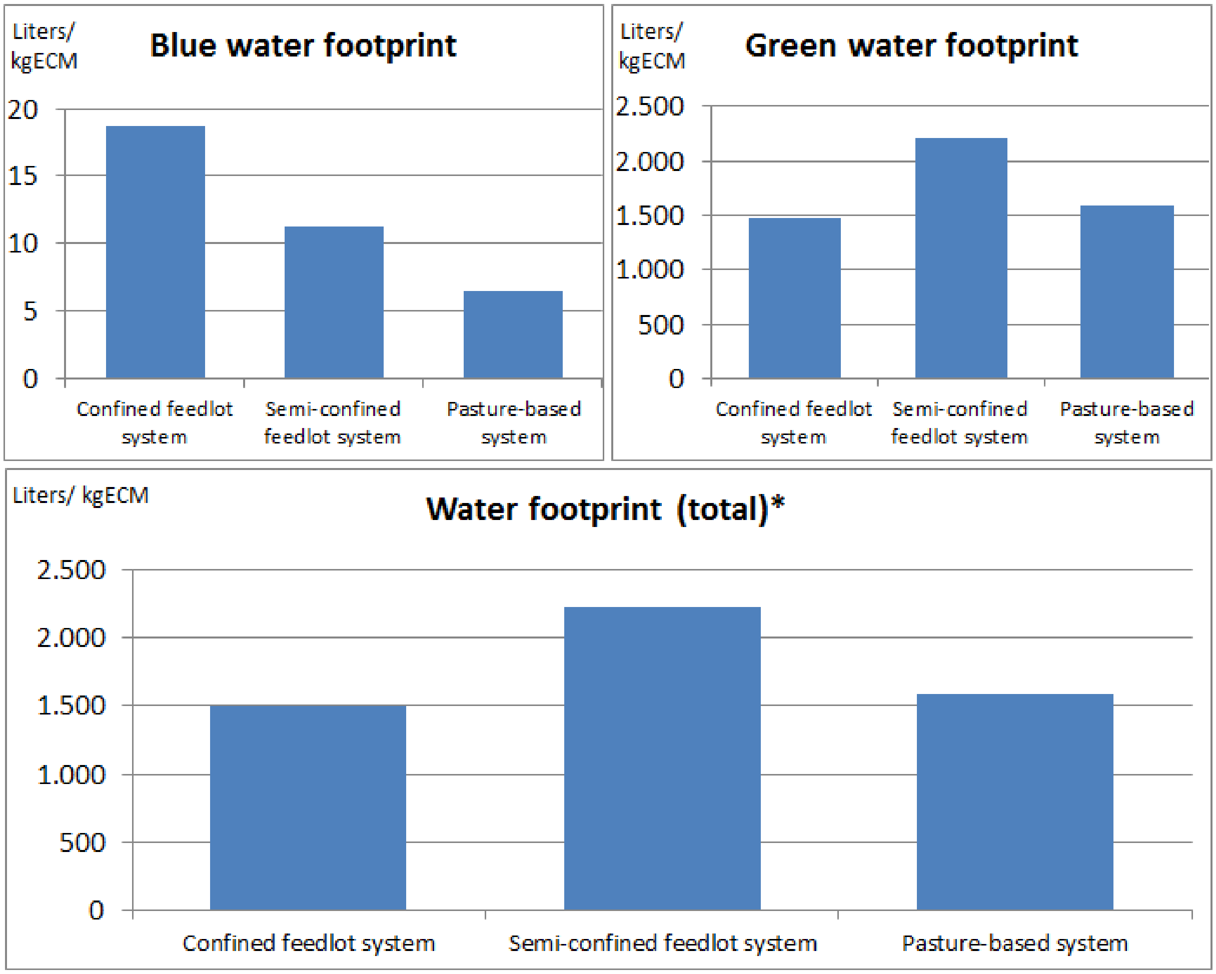
For green water we accounted the water content of the feed ingredients and grass (from grazing), based on data from literature (Brehmer 2008; Phyllis 2011). We also accounted for the water evapotranspired by these feed ingredients and grass (from grazing), also based the data on several other studies, which were from other areas of Brazil due to lack of available data and also lack of information on the origin of the feed ingredients: (1) For maize the evapotranspiration data was based on Albuquerque and Resende (2009); (2) for grazing the evapotranspiration data was based on Uda (2012); (3) for soybean the evapotranspiration data was based on Oliveira et al. (2011); for wheat the evapotranspiration data was based on Paiva et al. (2011); (4) for barley the data was based on Rodrigues et al. (2005); and for cotton the data was based on Pereira et al. (2013).

Even though the water footprint also considers gray water, we preferred to focus on green and blue water in this study, since the two latter represent the water consumed, the aspect that lacks on Léis (2013). Gray water represents rather an emission, and we think that other impact categories in the LCA study from Léis (2013) can better represent environmental impacts (e.g. eutrophication potential).

3. Results and Discussion

The results showed that milk from the confined feedlot, semi-confined feedlot, and pasture-based systems had blue WF of 19, 11, and 7 liters/kg ECM, respectively, and green WF of 1478, 2209, and 1584 liters/kg ECM, respectively (Figure 1). These results showed that semi-confined feedlot milk had the highest total WF, due to higher green WF results. This was mainly due to the high amount of brewers spent grain (byproduct of the beer industry) and maize in the feed composition.

Considering that blue WF is the traditional way to account for water use in LCA, it is interesting to note that even though confined feedlot milk had the highest blue WF, the bottleneck on (total) WF was found in the green water, with much higher results than blue WF. Therefore, a traditional LCA could have pointed out different conclusions from WF, as showed in this study (in traditional LCA confined feedlot milk would be the worst system for water use, while in WF system the semi-confined feedlot is the worst system).



* Except for gray water footprint

Figure 1. Blue water footprint, green water footprint, and the total water footprint for three milk production systems from southern region of Brazil.

Mekonnen and Hoekstra (2010) accounted for the green, blue and gray WF of several animal products from several countries. For milk produced in Brazil, the values ranged from 22-42 liters/kg of milk for blue WF, while for green WF the values ranged from 1046-1254 liters/kg of milk. It is possible to observe that the values for blue WF are similar to our system A, although it can be considered much higher when compared to our systems B and C. On the other hand, the green WF from our research (the three systems) are higher than from the values presented in Mekonnen and Hoekstra (2010). Our system A and C had values up to 51% higher, while system B had values up to 111% higher. The reasons for these discrepancies might be differences in amount of feed consumed (blue WF), percentages of feed consumed from irrigated systems (blue WF), amount of grazed land area (green WF), but also differences in evapotranspiration values, differences in the system boundaries considered, uncertainties on data collection, among others. The values presented in Mekonnen and Hoekstra (2012) and Mekonnen and Hoekstra (2010) for global average WF were between 790 and 1087 liters/kg of milk, for green WF, and between 49 and 82 for blue WF. Comparing our values with World average, we can see that blue WF were lower in our research, while our systems had higher values for green WF. The reasons for these differences can be the same as presented before, but for green WF it may also be due to climatic differences (e.g. higher temperatures in Brazil cause higher evapotranspirations).

The expansion of the system boundaries seems to be an important step on the WF methodology. For instance, the system boundaries for background blue WF can be exhaustive, if using life cycle inventory (LCI) databases (as ecoinvent database). On the other hand, the system boundaries have to be consistent among the different WF, and so far there is no LCI database that provides data on green or gray WF, most probably due to the complexity and local specificities of their calculations.

4. Conclusion

In this work we estimated the blue and green WF of three milk production systems in southern region of Brazil. Even though most of the data used to calculate the WF was based on secondary data, it was possible to observe possible hotspots (increase productivities of grains used in feed and/or feed conversion ratio), and to identify the systems with lower WF.

This was one of the first studies that published data on WF of milk production systems in Brazil. In comparison to other data from literature, we could observe that our values were higher for green WF, while blue WF had much lower results. These differences could be due to low amount of irrigated crop production systems considered, due to uncertainties on data collection, among other reasons.

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Assessing the land use impacts of agricultural practices on ecosystems

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ABSTRACT

In Life Cycle Assessment, agricultural fields have been defined as part of the technosphere. In order to cover environmental damage to fields during crop occupation and assess their reversibility, the different agricultural practices need to be covered by the land use impact category. The main goal of our contribution is to test the applicability of ongoing land use methods to assess agricultural practices, as well as provide recommendations for further research. Among existing complementary proposals of impact characterization models, we have chosen those, which could provide spatially explicit information on biodiversity damage expressed as species richness (relative or absolute number) and ecosystem services represented by net primary production depletion or biotic production potential. We have applied them in a case study of corn production comparing extensive and intensive management. Improvement of characterization factors for different land use intensity types and better definition of boundaries are main needs for further research.

Keywords: Biodiversity, Biotic Production Potential, Erosion, Organic matter, Technosphere

1. Introduction

In Life Cycle Assessment (LCA) agricultural studies, cropland is usually defined as part of the technosphere, and agricultural practices are assessed in relation to their effects on the surrounding environment (e.g. ecotoxicity, eutrophication). However, crop management affects the cropland itself as a natural resource. It can be argued that good management is reflected in the yields obtained, the quality of the crop and of subsequent ones. But a longer term vision, as well as the recognition of the importance of soil as provider of ecosystem services is necessary. Therefore, damage to the technosphere by the different agricultural practices needs to be covered by the land use impact category (or reconsider agricultural fields as part of ecosphere).

Land use impact assessment is a complex issue due to region specificities as well as the different nature of damages involved. The International Reference Life Cycle Data System Handbook (EU-JRC-IES 2011) and the ENVIFOOD Protocol (Food SCP-RT 2013) recommend cautiously (level III) the method that considers soil organic matter (SOM) (Milà i Canals et al. 2007a), because it is the most appropriate soil-quality indicator among the existing approaches to assess land-use impacts at midpoint level. No method was recommended for use at endpoint level; ReCiPe method can be used as an interim method.

Nowadays a lot of work is being developed in order to provide operational and globally-applicable methods to asses land use impacts in the framework of LCA. Milà i Canals et al (2007b), proposed that land use impact category should be assessed in terms of impact on biodiversity, impact of biotic production and impact on the regulating functions of natural environment. Within the framework of UNEP/SETAC Life Cycle Initiative a flagship project has been launched in 2012 to provide guidance and create consensus for assessing, among other LCIA indicators, land use impacts on biodiversity (Jolliet et al., 2014).

An updated version of the soil organic matter approach allows to use the change in soil organic carbon as an indicator for impacts on biotic production potential (BPP) (Brandão and Milà i Canals 2013), which in fact can also be considered as a measure of supporting services of ecosystems. Similarly Núñez et al (2013) developed a method using a growth-based value potential: net primary production depletion (NPPD) to assess the effect of soil erosion on ecosystem quality.

De Baan et al (2013) developed regional Characterization Factors (CFs) for land-use occupation and transformation using the species-area relationship model to assess the number of species that might be driven to extinction due to different land use types including agriculture.

More recently Elshout et al (2014) derived CFs based on findings of previous studies to assess agricultural land occupation on relative species richness. They provided midpoint CFs for different crops and differentiated between conventionally and low-input managed crops.

The main goal of our contribution is to test the applicability of ongoing methods to assess different agricultural practices focusing on endpoint damage, as well as provide recommendations for further research. We have applied them in a case study developed in the frame of a European LIFE project (F4F 2013-17), conducted in order to reduce the environmental consequences of manure livestock management.

2. Methods

2.1. Case Study

An LCA was carried out to compare intensive and extensive corn production. Crops were grown at the Mas Badia experimental station (La Tallada d'Empordà, Girona, north-east Spain) under different agricultural practices. Intensive corn production includes the use of chemical fertilizers and pesticides. Extensive corn production includes organic manure application and the use of catch crops as an intermediate crop to catch excess nitrogen and posterior treatment as a co-substrate in a biogas plant. In addition, growing catch crops are considered as a service to prevent erosion. We performed an attributional simplified LCA, where the functional unit was 1 ha. The life cycle stages included until farm-gate were: seeds production, manure, inorganic fertilizers, diesel consumption in labor operations and pesticide treatment. Livestock farming and posterior treatment for biogas plant are out of the scope of this study (they will be included in a subsequent step). In this paper we just focused on the land use assessment of foreground crops. A year period with one (intensive) or two annual crops, ryegrass and corn (extensive) are set as examples. Ryegrass was used as a catch crop previous to silage corn production. Table 1 shows relevant inventory flows of agricultural practices for each crop.

Table 1. Relevant inventory flows for ray-grass and corn.

LCI	**catch crop**	**corn extensive**	**corn intensive**
Seeds	26.70 kg·ha^{-1}	80000 u·ha^{-1}	80000 u·ha^{-1}
Date of sowing	Sept, 30th	Apr, 2nd	Apr, 2nd
Date of harvesting	Mar, 20th	Set,15th	Set,15th
Moisture content at harvesting	79.4%	78%	78%
Irrigation	--	2833 m^3·ha^{-1}	2833 m^3·ha^{-1}
Manure		170 kg·ha^{-1}	
N fertilizer	--		250 kg·ha^{-1}
P fertilizer	--		100 kg·ha^{-1}
K fertilizer	--		100 kg·ha^{-1}
Herbicide treatment	--		3.5 L·ha^{-1}
Tractor hours for different operations	7.7 h·ha^{-1}	12.6 h·ha^{-1}	12.6 h·ha^{-1}

2.2. Impact methods

Among existing complementary proposals of LCIA, we have chosen those for which inventory information was available and impact assessment provides information about agricultural impacts close to endpoint level for our specific area of study.

Following ILCD and Envifood protocol we used soil organic carbon (SOC), as a midpoint indicator and applied an updated approach (Brandao and Milà i Canals 2013) where the change in SOC is used as an indicator for impacts on BPP. Our area of study is located, according to this method, on the warm temperate dry climate region.

The effect of soil erosion on ecosystem quality is expressed using a growth-based value potential: net primary production depletion (NPPD). Indicator for soil erosion impacts was defined at the endpoint level for ecosystem quality in two steps: first relating soil loss to SOC loss and then relating SOC loss to ecosystem biomass productivity drop, NPPD. CFs are at a spatial resolution of 5 arcmin (Núñez et al 2013)

For biodiversity assessment the approach and CF for occupation and transformation provided by De Baan et al (2013) has been used. These authors calculated the total number of regional and non-endemic species lost per five different taxonomic groups (mammals, birds, plants, reptiles and amphibians) choosing biome units to derive CFs. This total regional damage was then allocated to the different land-use types according to the area share they occupied and their habitat quality. We applied the agricultural land use CFs for, Northeastern Spain and Southern France Mediterranean forest ecoregion, PA1215, (De Baan et al. 2013) for our case study.

In fact the area of study corresponds to. Climatically, the ecoregion experiences very hot and dry summers, and relatively temperate winters. Forests are mainly composed of mixed evergreen and deciduous broadleaf and conifer species. Endemism rate in the ecoregion ranges between 10-20% of the total vascular plants. Large mammals are not particularly prominent in this ecoregion. Most of the ecoregion has been intensively transformed into agricultural land or coastal urbanization for tourism.

Land use type according to Koellner et al (2013) corresponds to Arable Land, 5.1, differentiating between Arable, irrigated, intensive plus chemical–synthetic and organic fertilizer as well as pesticides use for corn (5.1.3.2) and Arable, irrigated extensive without use of chemical–synthetic or pesticides but indirect use of organic fertilizer and catch crop (5.1.3.1).

Table 2 shows main geographical characteristics of crop location relevant for the land use impact assessment methods applied.

Table 2. Geographical characteristics corresponding to location and each crop assessed.

Geographical characteristics	**Common to location**	**Extensive**	**Intensive**
Geographical coordinates	42º08' N; 2º99' E		
Climate region	Warm temperate		
Moisture regime	Dry		
Reference Ecoregion (De Baan et 2013)	Northeastern Spain and Southern France Mediterranean forests PA1215		
Land Use type (Koellner et al 2013)		5.1.3.1 Arable, irrigated, extensive	5.1.3.2 Arable, irrigated, intensive
Rainfall (mm y^{-1})	670.6	386.5	284.1
Sand (%)	64		
Silt (%)	23		
Clay (%)	13		
C org (%)		1.74	1.45
Land slope (%)	2		
Slope length (m)	100		
USLE ΔR-factor (MJ mm ha^{-1} h^{-1})	2360		
USLE K-factor (t ha h MJ^{-1} ha^{-1} mm^{-1})		0.026	0.027
USLE LS-factor (-)	0.285		
USLE C-factor (-)		0.475	0.755
USLE P-factor (-)	1		
Soil loss mass due to Erosion (t soil)		8.3	13.7

In addition, we provided the climate change impact for the same scope and reference period (one year of crop production to farm gate) expressed as kg CO_2 eq and in species·y following ReCiPe methodology to provide a reference to compare natural environment damage importance.

In order to have a detailed explanation of the methods, we invite readers to consult the corresponding references.

3. Results

The results shown in table 3 were calculated for the different approaches per ha and year. Despite the difficulties encountered to identify damage of the different land use intensity type, extensive production appeared to have lower impacts than their intensive counterparts. This is clear when applying BPP method, for which extensive practice did not report any impact. Considering uncertainty of results, none of the other methods can differentiate between different practices, regarding biodiversity damage (de Baan et al 2013), because there are common CFs for all agriculture types.

The use of catch crop helps to reduce soil loss in extensive systems and although the loss of organic matter is higher in intensive systems, there is not final difference in erosion ecosystem quality impact due to method applied to convert SOC loss in NPPD.

Transformation impacts are higher than occupation impacts. Land transformation impacts on BPP are one order of magnitude higher than land occupation impact on BPP. Biodiversity impacts due to land transformation were one or two orders of magnitude higher than occupation depending on the taxa.

Table 3. Results for the different impact categories are expressed in the respective units per ha and yr and show the geographical reference unit of CFs applied. Climate change is the full life cycle for an annual catch crop plus silage corn crops. Rest of the impact categories are related to one ha of land use for the foreground crops.

Damage Nature	**Units· $ha^{-1}·yr^{-1}$**	**Results Extensive**	**Intensive**	**Geographical unit**	**CF source**
Climate Change	kg CO_2 eq	1730	5300	Global	Goedkoop et al.2009
Climate Change	Species·yr	$1.37·10^{-5}$	$4.20·10^{-5}$	Global	Goedkoop et al.2009
Occupation					
SOC loss mass due to Erosion	kg SOC	144	199	Local	Núñez et al 2013
Erosion ecosystem quality	NPPD	2.66	2.66		Núñez et al 2013
Biotic Potential Production	kg C	no impact	$5.8·10^{3}$	Warm temperate dry region	Brandaö and Milà i Canals 2013
Biodiversity, Mammals	PLNE	$3.38·10^{-6}$	$3.38·10^{-6}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Birds	PLNE	$1.19·10^{-5}$	$1.19·10^{-5}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Plants	PLNE	$1.20·10^{-4}$	$1.20·10^{-4}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Amphibians	PLNE	$1.03·10^{-6}$	$1.03·10^{-6}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Reptiles	PLNE	$1.50·10^{-6}$	$1.50·10^{-6}$	Ecoregion PA1215	De Baan et al 2013
Transformation					
Biotic Potential Production	kg C	no impact	$4.5·10^{4}$	Warm temperate dry region	Brandaö and Milà i Canals 2013
Biodiversity, Mammals	PLNE	$1.82·10^{-4}$	$1.82·10^{-4}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Birds	PLNE	$7.51·10^{-4}$	$7.51·10^{-4}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Plants	PLNE	$4.34·10^{-3}$	$4.34·10^{-3}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Amphibians	PLNE	$7.59·10^{-5}$	$7.59·10^{-5}$	Ecoregion PA1215	De Baan et al 2013
Biodiversity, Reptiles	PLNE	$1.11·10^{-4}$	$1.11·10^{-4}$	Ecoregion PA1215	De Baan et al 2013

NPPD: net primary production depletion; PLNE: potentially lost non-endemic species;

4. Discussion

The different methods provide CFs based on homogeneous areas such as climate region (BPP) or biome (biodiversity damage). However, they are still large areas, which, depending on final goals of LCA, application can make comparisons difficult. From an applicability point of view, it is clear that more site-specific CFs for land use offer a better approach to grasping the local specificities of crop production.

On the other hand, it is necessary to develop CFs for more specific agricultural practice types (e.g., extensive, intensive, irrigated, greenhouse, organic agriculture) instead of generic ones (agriculture).

Main differences between both agricultural systems, extensive and intensive management are found in BPP land use impact assessment, because this category is using management factors from IPCC (2006), which give special importance to the carbon sequestration through the incorporation of manure (management factor 1.37 with manure vs 1.04 without manure). The use of management land use factors similar to those included in land use change could be potentially a good approach to consider the different agricultural practices.

Similarly, current erosion impact are applying different management factors related to agricultural practices and can provide information on long-term effects on soil resources but the revision and inclusion of more soil archetypes and conversion to biodiversity damage units are recommended.

Regarding biodiversity damage, vascular plants were shown to be the most sensitive group, followed by birds. This makes sense because usually plants are considered competitors of crops and removed from agricultural fields. These results are in agreement with the CFs derived by Elshout et al (2014). These authors differentiated between conventionally and low-input managed crops giving the respective CFs of 0.42 and 0.05 expressed as relative value of potentially disappeared fractions. Despite these results agree with previous studies, they shows different ratios depending on crops and area of study, which can confirm a tendency of lower impact for extensive systems but also the need for more accurate assessment methods.

Also according to the study of Elshout et al (2014), arthropods do not appear to be significantly impacted upon; however, we presume that the different management practices especially those related to pesticides use may have a large effect on them and therefore arthropods may serve as a good indicator for the impacts of differentiate practices.

Although regeneration times applied according to SOC indicator are lower than for biodiversity indicators, differences in importance between methods for transformation related to occupation can be explained because while de Baan et al (2013) calculated the transformation impact as a multiplication of occupation CF with half the regeneration time, Brandão and Milà i Canals (2013) included in this impact, the deficit of SOC due to the postponed regeneration of the system.

In this paper we have focused on crops (foreground system), issues related to land use on background processes have already been established in Milà i Canals et al (2013) and De Baan et al (2013).

5. Conclusion

Although it seems that extensive agricultural practices should mean lower impact when compared to more intensive practices, currently we have not enough developed impact characterization methods to assess and compare different agriculture intensities.

Among the different needs for further research, from our case study in particular, and agriculture in general, special attention should be paid to developing CFs for different land use agricultural types, as well as a better knowledge of relevance of the different taxa affected.

Together with the development and improvement of methods we would like to highlight the importance of testing them in practical and different case studies.

And last but not least a clear definition of boundaries, not only between technosphere and ecosphere, especially important in agriculture, but also which level of detail could/need to be covered by LCA. That means assuming the degree of implicit uncertainty, or otherwise the need to advise that environmental damage may/should/shall be assessed by other tools.

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Unit process data collection for specialty crop production

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Abstract

The United States Department of Agriculture (USDA) collects and releases a substantial amount of commodity crop production data to inform market activities. These data are frequently transformed into unit process data, which are used in life cycle assessments of agricultural production or agriculturally derived products. The USDA has recently transformed its Agricultural Resource Management Survey (ARMS) data into high quality and transparent unit processes that represent United States production for nine commodity crops in the USDA Life Cycle Assessment (LCA) Commons. However, neither high quality and transparent unit process data, nor complete input and yield data for specialty crops exist. State enterprise crop budgets are a consistent data source for specialty crop production and have been used in past agricultural LCAs (Adom et al. 2012, Matlock et al. 2008 and Eranki and Dale 2010). One of the major objectives of this project was to determine the reliability of "state enterprise crop budgets" as tools for collecting unit process data for crop production. This paper evaluates the applicability of state cost of production (COP) estimates, which underlie state enterprise crop budgets, to LCA.

State extension agricultural economists compile enterprise crop budgets from surveys of horticultural scientists, agricultural suppliers, marketers, crop consultants, and other local and regional stakeholders. The crop budgets define a range of management practices that vary based on local conditions and estimate the costs and revenues associated with production. Crop budgets are pragmatic and based on survey results from industry experts. However, crop budget survey methods are not necessarily standardized and may not be a reliable sampling of the industry. They can have a varying level of technological, temporal, and geographic specificity and vary across states such that state comparisons or multi-state aggregations may not be reliable. Therefore, indiscriminately transforming enterprise crop budget data into unit process data may not be scientifically defensible. Life cycle inventories calculated from corn production unit processes developed from the 2005 Iowa state enterprise crop budget for corn will be analyzed and compared to unit processes developed from USDA crop production surveys.

1. Introduction

The first objective of this work is to determine if "state enterprise crop budgets" and the "cost of production (COP) estimates," upon which they are based, can be reliably transformed into "unit process" data for strawberry crop production. We refer to gate-to-gate crop production as a unit process here, with a functional unit of a unit of crop produced, which is a part of larger product systems not treated in this paper.

In 2012, USDA-National Agricultural Library (NAL) released the *Life-Cycle Assessment Commons,* an open source, life-cycle inventory database for nine (9) commodity crops. The main data source used for developing unit processes for commodity crops is the annual USDA Agricultural Resource Management Survey (ARMS) (U.S. Department of Agriculture, Economic Research Service 2013). ARMS data are derived from an annual, national, statistical survey of field-level farm practices sponsored jointly by USDA Economic Research Service (ERS) and the USDA National Agricultural Statistics Service (NASS). USDA-NAL has transformed ARMS and NASS Quick Stats data into high quality and transparent unit process data that represent United States commodity crop production and can be used in life cycle assessment of agriculturally derived products. Crop production survey data represented in ARMS covers land occupation and transformation from previous crops, seed use, irrigation, tillage, crop residue management, and the use of nutrients, manure, and pesticides. When these data are combined with NASS Quick Stats data representing field crop production for each ARMS crop-state-year combination, the basis for an LCA unit process data flow is created.

A comprehensive federal survey of production practices or cost of production does not exist for specialty crops. USDA-NAL is investigating the transformation of crop production practices data that underpin the variable cost estimates of state enterprise crop budgets into unit process data that can be used for life cycle assessment of specialty crops. However, when analysts transform existing data for unintended uses, they must fully understand how the intended use and collection methods relate to the new questions they are asking. ARMS data were appropriate for transformation into commodity crop unit process data that represent U.S. agriculture because they were developed from a statistical sampling that represented historical activities. Their state level resolution and historical perspective make these data suited for policy analysis more so than producer planning purposes. Conversely, state cost of production estimates are not necessarily statistical samples and are developed for planning purposes. Therefore, they may not be representative of typical production practices and may not be adequate for policy analysis.

2. Methods

The basis for this work is a qualitative literature review and meta-analysis of the methods behind state and federal COP data. Our hypothesis is that the state COP data that underpin crop enterprise budgets are representative of state level management practices. This hypothesis is being tested with a two phased approach. The first phase, documented here, is a qualitative review of COP estimate methods and their applicability to LCA. The second phase is a quantitative meta-analysis that will compare life cycle inventory results from unit process for corn production developed from USDA ARMS data that reside in the USDA LCA Commons and corn production unit processes developed from state COP data collected by varying methods. Phase two results will be documented in a separate paper and presented at LCA Food 2014.

3. Results

Inconsistencies between the methods used by land grant universities developing state level COP estimates and USDA's Farm Cost and Returns Survey (FCRS) methods surfaced during congressional deliberations on the 1990 Farm Bill. Policy makers and researchers found that the variety of methods used in collecting, calculating, and analyzing state and national cost of production estimates yielded inconsistent analytical results. Essentially, using COP data gathered on a local or regional level with the intended use of improving farm management were not necessarily transferrable for federal policy making. Libbin and Torrel compared data collected in the 1986 crop and livestock budgets for New Mexico with data collected through the FCRS in that year. They found tremendous discrepancies among the state and federal data mainly due to methodological differences in data collection (Libbin and Torell 1990). An exploration of the discrepancies in state and federal cost of production estimates were formalized in a volume of Costs and Returns for Agricultural Commodities (Ahearn and Vasavada 1992), with the over- arching recommendation that states and the USDA must collaborate to coordinate state and federal cost of production estimation methods. As a result, the Commodity Costs and Returns Estimation Handbook was developed and is hosted on the web by the USDA Natural Resource Conservation Service (NRCS). The "handbook's purpose is to gather in one place information on estimating costs of and returns to agricultural enterprises (American Agricultural Economics Association 2000). In addition to publishing the Handbook, these activities culminated in the evolution of FCRS into the Agricultural Resource Management Survey in 1996. Presumably, the Handbook produced by this process helped align the methods used in state and federal cost of production estimates for commodity crops to support more consistent analyses.

3.1. Transforming COP Data into Unit Process Data

Transforming state COP data into unit process data requires an understanding of both the intended uses and data collection methods of COP data. In the Costs and Returns for Agricultural Commodities (Ahearn and Vasavada 1992), House (1992) and Guedry caution that the intended use of the COP data should determine the data collection methods. Alternatively stated, data collection methods must support their ultimate use. Guedry asserts that the purpose for which COP estimates are used influences the manner in which they are collected and that [if they are going to be used for research] analysts should focus on the differences that exist in the data and the procedures used in developing the estimates, rather than the formats in which they are presented (Guedry 1992). This statement is particularly pertinent for analysts using data for purposes other than their intended purpose. In this case, cost data are not being collected and used for economic analysis, but rather existing cost data are being used for environmental analysis. Klonsky asserts that in developing COP estimates each institution's methods are constructed following an internally standardized set of methods, but the methods used to estimate cost vary from state to state (Klonsky 1992). These variable methods may include both the intended uses of the COP estimates and the sampling methods.

3.2. Intended Use of COP Estimate

COP estimates are generally used for three purposes 1) assisting farmers in planning and improving their management performance 2) policy analysis (Morehart, Johnson, and Shapouri 1992) and 3) research activities (Guedry 1992). Therefore, cost of production information is of interest to farmers, farm management specialists, policy makers and researchers (Ikerd 1992). Subsequently, the first question that should be asked when using these data for analytical purposes is, "what does the economic [or life cycle] analyst need from a cost and returns data set? (House 1992)." It is proposed here that the material and energy input and output data be used to model field scale environmental emissions. Because results could inform both growers in improving production practices and/or commercial and government policy decisions, it must be determined if state strawberry crop production estimates provide an adequate representation of state-wide production. Indeed, many state enterprise crop budgets include a disclaimer that production practices and their associated costs may not be representative of production practices and costs throughout the state. Furthermore, "one fundamental difference among enterprise budgets is that some are intended to reflect historical events and are summary for the current year. Others are intended to be projections of costs for the coming year (Klonsky 1992)." The strawberry state enterprise crop budgets used in this study indicate that they represent "typical" state level production.

Extension economists may want to develop Cost and Return (CAR) estimates that are representative of progressive, well-managed farms (rather than all farms) engaged in the selected enterprise because those estimates may be more useful in guiding potential producers. On the other hand, the United States Department of Agriculture (USDA) and others producing historical estimates generally want to include a broader geographic area and to target all farms engaged in the enterprise regardless of whether they are progressive or not (American Agricultural Economics Association 2000) Therefore, state enterprise crop budgets may represent "target yields" or "safety first" estimates that may not necessarily be representative of typical production (Schoney 1992). However, enterprise cost budgets are usually part of on-going and even long-term programs in which production practice and cost estimation methods have been refined over time. The data quality conferred by the longevity and expertise of the stakeholders involved in developing state cost of production estimates is certainly implicit, if difficult to measure.

3.3. Data Collection Methods

If the intended uses of the COP data collection yield data that are representative of crop production practices, the methods by which COP data are collected must also yield representative estimates. The most important factor in assuring data quality is statistical inference. Statistical inference determines whether, and to what extent, results from the analysis and estimation can be generalized to a broader set of farming operations. Statistical inference is largely determined by two activities: precisely defining the group (or target population) the analyst wants to investigate, project for, or draw conclusions about; and selecting representative data from that population for the analysis. There are two sampling methods that allow analysts to select a representative sample of data from the target population in such a way that valid inferences can be maintained: a statistical sample and a judgment sample (American Agricultural Economics Association 2000).

A statistical/probability sample is considered superior to a judgment/non-probability sample. A statistical/probability sample is one in which each farm in a targeted population has a positive and knowable chance of being included and produce data that are representative of the entire target population. Judgment samples, on the other hand, are selected through some method other than statistical sampling, usually the subjective decision of one or more individuals. Consequently, at least some members of the target population did not have a chance to be selected. Judgment surveys are not necessarily inaccurate, but it is also not possible to determine if they are accurate either. The accuracy of the sample depends on the expertise of the individual selecting the sample (Williams 1978). State COP estimates are usually derived from judgment samples of the local and regional industry. In rare cases, state COP estimates are derived from statistical samples for commodity crops. However, statistical sampling is cost prohibitive and it is not realistic to accomplish for each specialty crop or even each commodity crop in each state in every year.

In fact, statistical sampling may not be appropriate for specialty crop COP data collection. Judgment samples may produce more reliable data when the sample size is very small. Also, depending on the specialty crop and state, judgment samples may be more effective in terms of cost and reliability. Another disadvantage of probability surveys is that they are not collected longitudinally and lack richness over time. State COP estimates, on the other hand, “are designed to provide usable estimates within the resource constraints of those preparing the estimates. Information that is developed annually is, in general, part of an ongoing educational program that has been developed over a period of years. Such programs have an advantage of being validated annually by users (Guedry 1992).” The veracity of the data gathered through long-term judgment sampling is likely to continue to be improved over time and may be more representative of reality than statistical sampling.

4. Discussion

The results of this review indicate that transforming state enterprise crop budgets into reliable unit processes for life cycle assessment is feasible. The “static snapshot” of COP estimates for an agricultural enterprise is aligned with the nature of a unit process. However, several subtle issues with respect to the use of crop budget derived unit processes in life cycle assessment became apparent. Primarily, cost estimates cannot be categorically transformed into flow data and data collection methods vary among states and unit processes developed from state COP data must be used with caution in LCA. COP data collection methods must be understood, reconciled across states, and fully documented if the unit processes are used as part of an aggregation or comparative LCA. Aggregations or comparisons across states cannot be reliably made based on state COP estimates. Second, the data used to build state enterprise crop budgets do not necessarily produce “representative” production practice data. Before using an enterprise crop budget, the life cycle analyst must endeavor to understand how the data were collected and for what purpose. Unit processes for a single state is reliable if the crop budget states that production practices and costs are representative of state-wide production. A statistical sample is not essential for collecting reliable data, but the survey must be deemed to be representative of current activities.

Most state COP estimates for commodity crops are based on judgment samples because the cost of ongoing statistical sampling is cost prohibitive. State specialty crop markets cannot bear the costs of ongoing statistical sampling either. However, statistical sampling may not be appropriate for state specialty crop markets due to small sample sizes of producers in each market. The ongoing, annual sampling of the industry and continuous improvement of methods and estimates provides confidence in the quality of COP data collection methods. State economists sample the local and regional industry over a period of years and continuously endeavor to improve their methods and estimates. The institutional knowledge and continuous improvement developed over a period of years in these programs provides confidence that state COP estimates and their underlying production practices do, indeed, represent state production. These data may be more reliable than a statistical sample of the same market.

5. Conclusion

While data collection methods for state enterprise crop budgets are reliable and internally consistent within a state from year to year, survey methods vary among states. Therefore, budgets must be individually evaluated to determine if they are reliable for comparative LCA. Unit processes can be reliably developed for individual states, but differences in data collection methods among states must be reconciled before the unit processes are used in LCA. If state enterprise crop budgets as a whole are to be reliable data sources for LCA, data collection methods must be standardized beyond format. State enterprise crop budgets are designed for field-level micro-economic planning and may not provide the resolution of input/output data necessary for a detailed field-level LCA. For example, a budget may estimate an aggregated cost for pesticides required for a crop in that year, but may not define which specific pesticides or the quantity of those pesticides used. Crop budgets may be more appropriate for developing background unit process data for more complex agriculturally derived products rather than higher resolution data required to inform field level activities. In the absence of standard methods across states, the analyst must understand and reconcile variability among states before making comparisons and the intended use of the budget. The variability among state budgets and their intended application must be fully transparent in the LCA documentation.

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The applicability of LCA to evaluate the key environmental challenges in food supply chains

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ABSTRACT

System analysis was performed to gain an overview of key environmental challenges and pinpoint hotspots causing environmental impacts in three European food supply chains. An overview was obtained based on a review on LCA studies for beef, dairy, orange juice and aquaculture food supply chains. Similarities of the main environmental impacts were identified to rationalize and justify the selection of key performance indicators chosen for a simplified web based LCA tool developed within the EC funded project SENSE (FP7). Life Cycle Assessment methodology covered many of the key challenges identified but will not be sufficient to address all environmental impacts generated from the food supply chains. Especially for aquaculture impacts that are not taken into account with LCA are i.e. nutrient and organic matter releases, impacts associated with feed provision, diseases introduction, escapes, and changed usage of coastal areas. In agriculture land use and biodiversity are issues not well covered.

Keywords: Environmental challenges, agriculture, aquaculture, LCA, key environmental performance indicators

1. Introduction

The food and drink sector is the largest manufacturing sector in the European Union (15%) which directly employs 4.1 million people but it is a highly fragmented sector consisting of 99% of SME's and microenterprises (FoodDrink Europe, 2011) which makes the supply chain from "farm to fork" complex with a multitude of actors involved. This sector is also associated with high environmental impact; a study commissioned by the European commission showed that three areas of consumption – food and drink, private transportation, and housing – are together responsible for 70-80 % of the environmental impacts of private consumption. Food and drink consumption was responsible for 20- 30% of the various environmental impacts of total consumption, and in the case of eutrophication for even more than 50% (Tukker, 2005). Meat products have generally a higher environmental impact than vegetables and fruit (Mogensen *et al.*, 2009; Figure 1) but variations within product groups can be highly depending on production system. Geographical region also influences the environmental sustainability.

Life cycle assessment (LCA) according to an ISO standardized method (ISO 14040, 2006) is generally applied to quantify the environmental impact for a product or service from cradle to grave. LCA is a valuable method but it is difficult to compare one food product to another due to differences in the goals and scopes of the different studies. Defra recently reviewed over 180 LCA studies on food and found that most studies end at the farm gate of the food supply chain, and very few extend as far as the consumer and that there is a shortage of studies which include information on distribution, retail and consumption and waste (Defra, 2011). Due to differences in goals and scopes, methodological approaches, setting of boundaries and assumptions as well as data quality and availability it is often not possible to compare the environmental impact of food products within the same product category and even less so between different product groups.

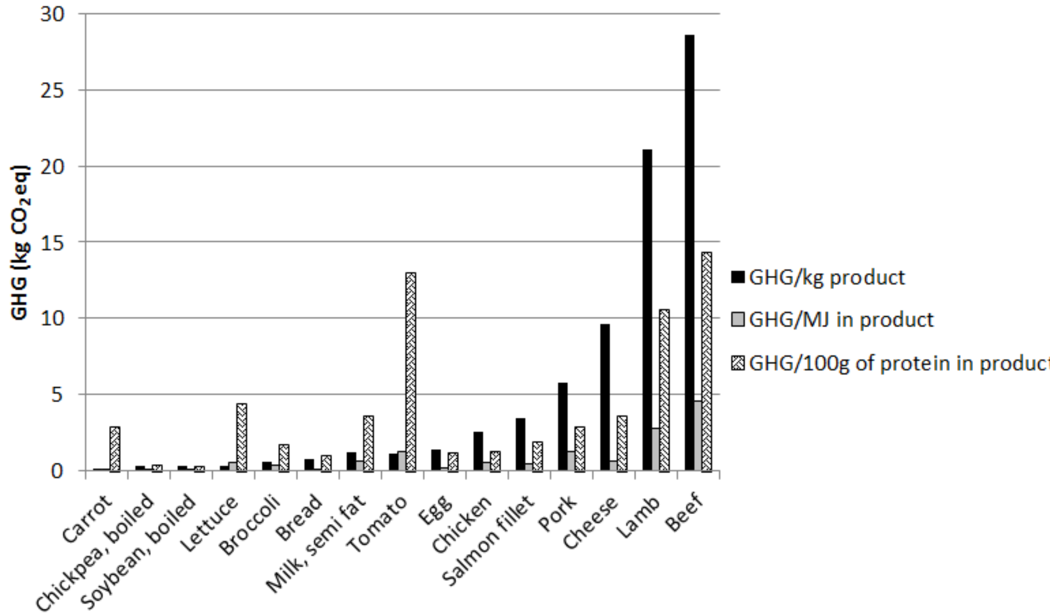

Figure 1. Greenhouse gas emissions of a range of food products per kg of product, per energy content and protein content from farm to retail. The figure is based on the following references: vegetables- Sonesson *et al*. 2010 except chickpea from Gan *et al*. (2011) and soybean from Prudencio da Silva *et al*. (2010); meats (bone free) Cederberg *et al.* (2009) and dairy products Flysjö, (2012). Data on protein content is from the Swedish Food Agency (www.slv.se).

In the SENSE project the aim is to obtain a simplified impact assessment, thus the most relevant impacts that are common in all food supply chains are chosen in order to design a simple LCA based tool, the SENSE-tool. As a first step to gain an overview of challenges in food supply chains in Europe a literature review of existing LCA studies was performed to identify the key environmental impacts as a basis to select key performance indicators that can be used in a simple life cycle assessment tool.

2. Methods

Key environmental challenges in food supply chains were identified based on literature review by selecting as case studies the following food sectors to represent a range of variation of food supplied to the European market:

- Orange juice (Esturo *et al.*, 2013)
- Beef meat and milk (Aronsson *et al*., 2013)
- Aquaculture (salmon) (Olafsdóttir *et al*., 2013)

Published life cycle assessments (LCAs) were chosen as the preferred source of evidence for quantifying environmental attributes of the selected food chains. This is not an exhaustive literature study but key references have been chosen to gain an overview and to compare the impacts in the different food supply chains. The main aim is to demonstrate the similarity and variability of the environmental impacts by giving a range of values reported for the most common environmental impacts assessed by LCA in the respective food supply chains.

3. Results- Environmental challenges

In general supply chain of food products share similarities where food moves from producer to consumer via the processes of production, processing, distribution, retailing and consumption. Each step of the supply chain results in an environmental impact on local, regional or global level (Figure 2 and Table 1). If the environmental hot spots or a key environmental challenge for each step in the chain is known, efforts can be used to focus on these problems and effectively minimize the environmental impact.

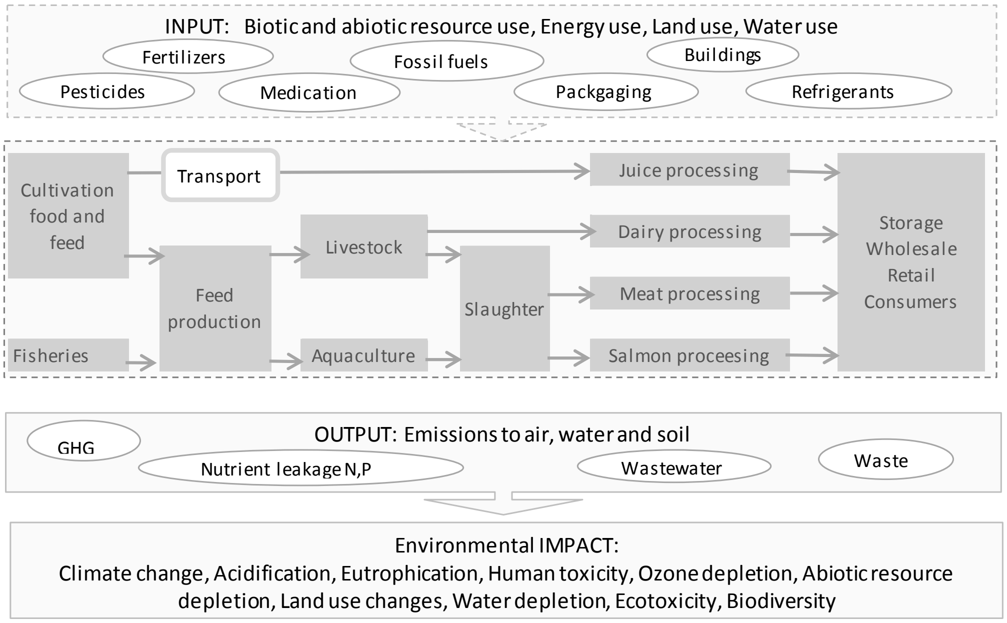

Figure 2. A simplified overview of the common steps in three food supply chain systems (orange juice, meat/dairy and aquaculture) and examples of input resources and output emissions causing environmental impacts.

The environmental challenges of food supply chain systems are typically caused by the use of fossil fuels, water, fertilizers, pesticides and land in the primary production as well as the accumulation of discards from the farm animals i.e. manure and excessive feed in aquaculture. Other challenges in farming are related to the use of feed, medication, the occurrence of escapees and the impacts of the farming on the wild flora and fauna. In the processing steps of food the use of resources, energy, water, packaging and refrigerants, waste and wastewater generation, and the type of facilities like buildings and infrastructure will influence the overall environmental impacts. The choice of technologies is often a key influential factor and the yield, full utilization of resources including the use of by-products is of importance in the overall estimation of the environmental load both in production and processing. In the transport phase the use of diesel is a challenge where the mode of transport can influence the severity of environmental impacts.

Table 1. Identified key environmental categories for the food chain, relevant production steps and challenges.

Impact category	Production step	Challenges
Climate change	Cultivation of biomass, Fishing (feed fish), Feed industry, Animal rearing, Food industry, Transport	Nitrogen fertilizer production, Use of N fertilizer and manure, Enteric fermentation, Manure handling, Use of fossil fuel, Refrigerant leakage
Acidification	Cultivation of biomass, Fishing (feed fish), Feed industry, Animal rearing, Food industry, Transport	Use of N fertilizer and manure, Use of fossil fuels
Eutrophication	Cultivation of biomass, Fishing (feed fish), Feed industry, Animal rearing, Food industry, Transport	Use of N and P fertilizer and manure, Manure handling, Nutrient release from aquaculture, Waste water
Human toxicity	Cultivation of biomass	Use of pesticides
Ecotoxicity	Cultivation of biomass, Feed industry (fishing), Animal rearing, Aquaculture	Heavy metals from fertilizers, Use of pesticides, Use of medicines, Use of antifouling paint
Land use	Cultivation of biomass, Animal rearing (Aquaculture sea surface/floor use)	Land use efficiency
Abiotic resource depletion	Cultivation of biomass, Fishing (feed fish), Feed industry, Food industry, Transport	Use of fossil fuels, Use of P fertilizer
Water depletion	Feed industry, Food industry, Transport	Irrigation, water use
Biodiversity	Cultivation of biomass, Animal rearing	Land use, Use of pesticides, Escapes and diseases in aquaculture

3.1. The beef and milk supply chain

The farm stage is generally the main contributor to the total environmental impact of a food product and for beef and dairy chain the farm stage contributes approximately 80 to 95% percent to the total emissions (e.g. FAO, 2010), with grassland systems having higher emissions than mixed systems (FAO, 2010). Methane is the main contributor with an average of 52 percent of the total emissions. CH_4 emissions from ruminant livestock systems come from enteric fermentation that ruminants produce as a by-product of their metabolism (Crosson et al., 2011) and approximately 6 percent of the energy intake converts to CH_4 but can vary between 2-12 percent depending on the diet. The second largest contributor to environmental impact from beef and dairy systems is N_2O emissions (approx. 27%; FAO, 2010) which can arise as direct emissions from organic manures or inorganic fertilizers applied to soil. In addition, indirect N_2O emissions associated with agriculture arise from volatilization of land applied manures and/or N based fertilizers, and N lost via runoff and leaching from agricultural soils (IPCC, 2006). N_2O emissions also occur upstream when mineral fertilizers are produced. Besides being a potent GHG, N_2O also contributes to ozone depletion. Carbon dioxide emissions contributing to climate change potential and acidification are caused by energy use from operations using fossil fuels and also from production of mineral fertilizers.

There are many LCA studies on beef and dairy production up to farm gate but it is rare with studies that follow the beef and milk to the consumer or grave. Post farm stages for milk and beef production has been estimated to 5-20% of the total environmental impact depending on what life cycle stages are included (Sevenster and de Jong, 2008). Energy use during the processing of milk and the packaging have major influence post farm. Transport often has minor influence but it depends on mode of transport and distances. The processing in the dairy plant results in the highest environmental impact. The separation, homogenization and pasteurization use the most energy (Nilsson and Lorentzon, 1999). However, the cleaning operations have also been identified as a major source of environmental impact (Hogaas Eide, 2003).

Packaging material is important for the environmental impact. Carton packs for UHT milk have been shown to have a significantly better environmental profile compared to HDPE and PET bottles with respect to CO_2 emission, use of fossil resources and consumption of primary energy. Recycling beverage packaging materials as a lower environmental impact than disposal in landfills or incineration plants (Meneses *et al.,* 2012). Up to 17 percent of the total life cycle emissions for energy and 18 percent for global warming potential can be contributed to the manufacturing and distribution of milk packaging consisting of paperboard carton (Hogaas Eide, 2003). The waste management of packaging can contribute significantly to the environmental impact. Packaging design is also an important factor when it comes to product loss in the consumer phase.

No information has been obtained when it comes to the environmental impact of beef and dairy at retail.

At the consumer stage the energy used to refrigerate milk and meat will result in the major environmental impact. There can be large variations in energy consumption of household refrigerators with up to 11.27 MJ/liter and year difference between the lowest and highest energy consuming refrigerator (Sonesson *et al*., 2003). To prepare the meat also demands energy. When it comes to waste in the household there are different kind of losses of milk and meat. In the milk and dairy chain approximately 11 percent of the milk is lost due to wastage (Gustavsson *et al*., 2011). For the meat chain (note, all meats) the wastage is around 20 percent. The consumer stage is the main stage were wastage occur for milk, dairy and meat products (Gustavsson *et al*., 2011).

3.2. The orange juice supply chain

During the cultivation of orange trees high fertilizer rates and large applications of manure concentrated in specific geographical areas lead to significant emissions of ammonia and nitrate, which creates eutrophication and acidification in sensitive aquatic and terrestrial environments and pollution of ground and surface water (EEA, 2009). The use of pesticides affects toxicity both for humans and the ecosystems. Eutrophication caused by agricultural practices has been identified as a serious problem in several Mediterranean countries. Specifically, Comunidad Valenciana region, which is the main orange production area in Spain, is included in the vulnerable zones to nitrate pollution from agricultural sources. Nevertheless, Murcia region and the West of Andalucía are also important orange producing areas. Moreover water depletion effect may be severe in Mediterranean area, where water scarcity is a particularly significant problem. The loss of biodiversity, changes to biotic interactions, and resource depletion in ecosystems are possible impacts of water scarcity.

The main contributor to atmospheric emissions at the processing stage is the energy consumption, mainly for thermal treatments (i.e. pasteurization, evaporation) which is usually related to the combustion in boilers of fossil fuels. Cooling and refrigerated storage of juice have also electricity requirements that implies indirect environmental impacts. Additionally, leakage of refrigerants fluids can be a direct source of GHG emissions contributing to climate change and ozone layer depletion potential. Carton based packaging dominates with 60% of total volume consumed and this type of packaging has been shown to have the lowest environmental impact concerning climate change, energy use and eutrophication when compared to glass and plastic (PET) packaging (Labouze, *et al*, 2008). For the retail and consumption steps of the orange juice value chain very little environmental information exists but it may be concluded that the impact of these steps are small. The main contributors to the environmental impact from the orange juice supply chain are fuel consumption during transport or distribution, followed by the use of fertilisers, herbicides and pesticides and consumption of fossil fuels in agriculture (Beccali *et al*, 2009). Regarding eutrophication potential more than 80% of the total impact is due to the use and production of fertilizers, pesticides and herbicides. The transport and agricultural inputs are the most influential stages of the orange juice value chain.

3.3. The salmon aquaculture supply chain

Environmental challenges in aquaculture which have been in focus are related to the potential loss of biodiversity due to the use of medicine for control of diseases and the salmon lice, and the effect of escapees on the wild salmon. The efficiency of feed and farming systems can have an impact e.g. excess feeding causes eutrophication which influences the benthic ecosystem because of nutrient enrichment of sediments and the water column. Exploitation of forage fish for feed has been a controversial issue, since this puts pressure on fish stocks and may have an impact on the seafloor. The use of soya and other crops for feed has also a considerable environmental impact because of land use changes caused by cultivation and the use of fertilizers, pesticides and water for irrigation. These issues have influenced public opinion and their perception towards aquaculture, which is sometimes a priori negative image.

The salmon production industry is concentrated in Northern Europe, Canada, and Chile and studies on LCA of aquaculture of salmon have focused on the effects of different composition of feed in these countries. While net-pen systems are most common for salmon LCA studies are also available for other rearing technologies like closed system aquaculture and other species farmed in Europe like trout, and arctic charr, as well as turbot and seabass.

Feed production is most often the major contributor to environmental impacts in conventional aquaculture systems (Ellingsen and Aanondsen, 2006; Ziegler *et al.*, 2012), while the impact of energy use is dominating in recirculation systems (Ayer and Tyedmers, 2009). Feed producers source raw materials from diverse fish, crop, and livestock sources globally, each with characteristic resource dependencies and environmental impacts. Fuel use in fishing, and feed production in aquaculture are key contributors to greenhouse gas emission (Ziegler *et al.*, 2012) and the impact of fuel use for global transport involved in sourcing feed is also of concern.

Water use is of importance especially in water scarce areas and land based systems and water use for irrigation in production of crop for feed.

Processing, packaging, transport, sale, consumption and waste management have not been commonly included in life cycle stages in seafood LCAs. This is particularly the case in aquaculture studies, while fisheries studies have often followed products through the transport stage (Ziegler *et al.*, 2012). Results from recent studies which have focused on environmental impacts of the processing and transport steps have shown that they are not significant in the overall impacts for the products when the transport is a short distance to the market within Europe. However, when considering the product type (whole fish or fillets), long distance transport and mode of transport (air or ship) the transport was found to have a large impact on the energy use and GWP and trucking is also an important contributor to GWP. LCAs that have focused on the transportation phase of chilled fish supply chains agree that sea freight is by far more environmentally friendly transportation mode than air freight and therefore it is very important to consider how food is produced and transported to the market and not only where it is produced in terms of environmental performance of products (e.g. Tyedmers *et al.,* 2010; Ingólfsdóttir *et al.*, 2010).

The impacts of packaging material and chilling in transportation were the main contributors to environmental impact potentials in a seafood supply chain systems (not including the fisheries) when comparison was made between chilled and superchilled fillets (Claussen *et al.*, 2011). The environmental impact of EPS packaging has been shown to be considerable, where the main contribution is energy use in the production of EPS granulates (Ingólfsdóttir *et al*., 2010).

The impact category ozone depletion is related to the use of refrigerants. However, new refrigerants are being developed where replacement of the HCFC R22 with environmentally harmless refrigerants like ammonia is in progress. According to Ziegler *et al.*, (2012) this change would reduce the carbon footprint of fish products by up to 30% if the right substitutes were chosen.

The aquaculture sector has made considerable efforts to mitigate the environmental effects for example by changing the composition of feed and development of aqua feed, as well as improved aquaculture technologies and good practices. Governmental monitoring and legal requirements in many countries require that aquaculture farms report occurrences of sea lice, escapees, the use of medication and water quality and sediment monitoring in the areas close to the farms. This implies that data on these aspects may be readily available and currently there is an increasing awareness that monitoring data should be accessible in the public domain to enhance the transparency and help building an image of responsibility for the sector.

Table 3 shows reported values from LCA studies for four of the most common impact categories for milk, beef orange juice and salmon aquaculture. Much of the variation in the values is due to differences in goal and scope, if the studies were attributional or consequential or how the metrics were reported (especially for energy use).

Table 3. Examples of reported key values for four impact categories for milk, beef, orange juice and salmon aquaculture.

Impact category	Milk[a]	Beef[b]	Orange juice[c]	Salmon Aquaculture[d]
Climate change (kgCO_2e/kg)	0.5-1.8	20-27	84-112	1.8 – 3.3
Acidification (g SO_2e/kg)	4-7.5	101-510	0,.5-5.5	10.3 -29.7
Eutrophication (g PO^{4-}/kg)	4.8-19	59.2-169	2.5-11.3	31.8 – 74.9
Energy use (MJ/kg)	1-6.9	325-1650	764-952	26,2 – 32,1

[a]Milk: Cederberg, Flysjö and Mattson (2007), Thomassen *et al*. (2008), Meneses *et al.* (2012);
[b]Beef: Casey and Holden (2006), Cederberg *et al*. (2009), Nguyen *et al*. (2010);
[c]Orange juice: Knudsen (2010), Sanjuán (2005), Ribal (2009);
[d]Aquaculture: Pelletier and Tyedmers (2007) Ayer and Tyedmers (2009) Pelletier *et al*. (2009) Ellingsen and Aanondsen (2006) Boissy, et al. 2011.

4. Discussion

4.1. Issues not covered by LCA

There are other aspects that so far are not adequately included in LCA's studies due to lack of consensus methodology (e.g. soil carbon dynamics) and data availability (e.g. waste percentages). As LCA is a method for assessing the environmental sustainability, socio-economic indicators (e.g. animal welfare) are typically not analyzed in LCA. The inclusion of social impacts will be considered in the SENSE project but these are typically associated with the performance of the enterprises, working conditions and employee rights.

Many of the aquaculture-related environmental impacts are not incorporated in appropriate impact categories in LCA and therefore, LCA methodology for environmental assessment will not to be sufficient to address all of the key global challenges generated from aquaculture i.e. nutrient and organic matter releases, impacts associated with provision of feed, introduction of diseases, introduction of exotic species, escapes, and changed usage of coastal areas (Samuel-Fitwi *et al*., 2012). The indicators and methods applied for chemical discharges and assessment of ecotoxicity are not well developed and their use for environmental impact assessment of aquaculture have been questioned (Ford *et al.*, 2012). Land use for crop production for feed and sea primary production - required to sustain the fish used for salmon feed and the benthic area influenced by fishing gear and methodologies have been suggested to calculate these effects to assess the impacts of feed for salmon (Ytrestöyl *et al.*, 2011). Therefore, when developing a simple tool for environmental assessment like the SENSE tool the limitation and justification for the LCA approach and the methodologies applied need to be addressed.

Some of those aspects that could be addressed in addition to using the LCA methodology are listed here:

- Amount of wastage could be used to measure efficiency in the supply chain. Food wastage reduction has been identified as an important part of EU policies over the last years.
- Feed efficiency in animal rearing can be a relevant indicator to evaluate several environmental impacts. For example, higher feed efficiency results in increased daily weight gain of cattle which means that rearing times can be reduced. This in turn can lower environmental impacts.
- The use of fish for feed is a biotic resource use for which no validated LCIA method yet exists. However, the use of forage fish and the FIFO (Fish in – Fish out ratio) could be used as a basis for an indicator.
- Carbon sequestration can be a positive effect from some systems, e.g. pasture based cattle production, while on the other hand annual cropping can lead to soil carbon emissions. Methods to measure carbon sequestration form different rearing system should be developed.
- Animal welfare is of concern and different opinions exist depending on culture and history in different countries. A production system for animals can be very efficient for the environment but result in poorer animal welfare.
- In some countries esthetical values in the countryside are discussed. Without any grazing animals the biotope will change from agricultural landscape to forests or larger monocultures of arable farming. This can have an impact on the rural communities both for inhabitants and visitors. In this case indicators for eco system services are being developed and could be considered.
- Use of antibiotics versus preventive actions to improve the animal health is very much discussed. Antibiotics can give higher and more efficient production as well as there is a risk for releases to the environment that can introduce resistance to the antibiotics. This is not integrated in LCA.
- How to assess the effects of land use change (LUC) and indirect land use change (ILUC) is very much discussed in the LCA community. There is not yet any consensus on how this should be integrated in LCA.
- Effects on biodiversity are often included in the land use impact assessment methods. It is widely recognized that it is important to assess the land use impact on biodiversity but because of the complexity of the issue there is no widely accepted methodology to use.

5. Conclusion

From the review of LCA studies the main environmental challenges for the three food supply chains have been highlighted and the most important issues can be assessed by LCA based methodologies and included in a simplified LCA web based tool. There are however environmental challenges that LCA methodology does not cover which are important to consider when assessing the environmental sustainability of a food supply chain, especially related to effects on biodiversity. Additionally there are other aspects and challenges that are not environmental, e.g. animal welfare and release of antibiotics to the environment, which are not possible to address with LCA methodology.

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Environmental impacts of producing bouchot mussels in Mont-Saint-Michel Bay (France) using LCA with emphasis on potential climate change and eutrophication

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ABSTRACT

LCA of bouchot blue mussel culture in Mont Saint-Michel Bay (France) was performed to identify impact hotspots of the activity. To better characterize potential positive effects on eutrophication and climate change, the chemical composition of the flesh and shell was analyzed. A small but potential mitigating effect on eutrophication was observed, reaching 1 kg PO_4-eq. per tonne of "ready-to-cook" mussels. The potential carbon sink effect is influenced by hypotheses about the wooden stakes of the bouchots and about the fate of the shells, associated to the management of discarded mussel and of the household waste. This effect barely compensates the climate change due to the use of fuel for on-site transportation. In addition, environmental impacts of blue mussel culture depend on its production location, as a function of mussel yields due to the marine currents combined with the distance to on-shore infrastructure.

Keywords: LCA, Mussel, Carbon sink, Eutrophication, Climate change

1. Introduction

Blue mussel (*Mytilus edulis*) culture yielded 207,918 tonnes of shellfish worldwide in 2010 (FAO 2012). Blue mussel culture in France represents about 35% of world production (CNC 2012) and is performed using two different techniques: long-line rafts and bouchots. Bouchot culture is currently the only type of mussel culture in Mont Saint-Michel Bay, which produces 10,000 tonnes of blue mussels annually. Bouchot culture consists of using wooden stakes sunk into the foreshore as a culture support for the mussels.

In the scientific literature, interactions between mussel culture and the environment have been studied mainly from a nutrient-cycling perspective (Brigolin et al. 2009, Jansen 2012, Nizzoli et al. 2011, Richard et al. 2006) or one assessing biodeposition and benthic effects (Chamberlain et al. 2000, Christenssen et al. 2003, Grant et al. 2012). Few authors have assessed environmental impacts of mussel culture with a broader viewpoint using holistic approaches. Thrane (2004) assessed energy consumption of mussel production in Denmark, Fry (2012) estimated a carbon footprint of longline mussel production in Scotland, and Iribarren et al. (2010a,b,c) performed Life Cycle Assessment (LCA) of the longline mussel supply chain in Spain. Among these studies, the methodology used to establish an inventory differs and the inclusion of effects of mussels on nutrient and carbon cycling in the marine environment varies.

Mont Saint-Michel Bay is located between Brittany and Normandy, two regions of high livestock production, where water quality is regularly called into question due to overloading of nitrogen (N) and phosphorus (P) caused by agricultural activities. Our study aimed to evaluate environmental impacts of bouchot mussel production in this context. In particular, we investigated its potential role in mitigating climate change by stocking carbon in mussel shells and potential eutrophication by extracting nutrients from the bay.

2. Methods

2.1. Goal and scope

LCA was performed to assess environmental impacts of Mont Saint-Michel mussels produced according to the Appellation d'Origine Contrôlée (AOC) label, with the double objective of establishing an environmental profile of the activity and highlighting potential impact hotspots. The LCA followed the main guidelines of the ILCD handbook (JRC 2011), with a cradle-to-gate approach. Boundaries of the production system included spat collection and transportation, the culture stage (including equipment use), and the processing and packaging stage at the producer organization's plant (Fig. 1). The distribution, marketing and consumption stages are not included in the study. The functional unit was 1 tonne of packed, “ready-to-cook” mussels at the producers’ plant.

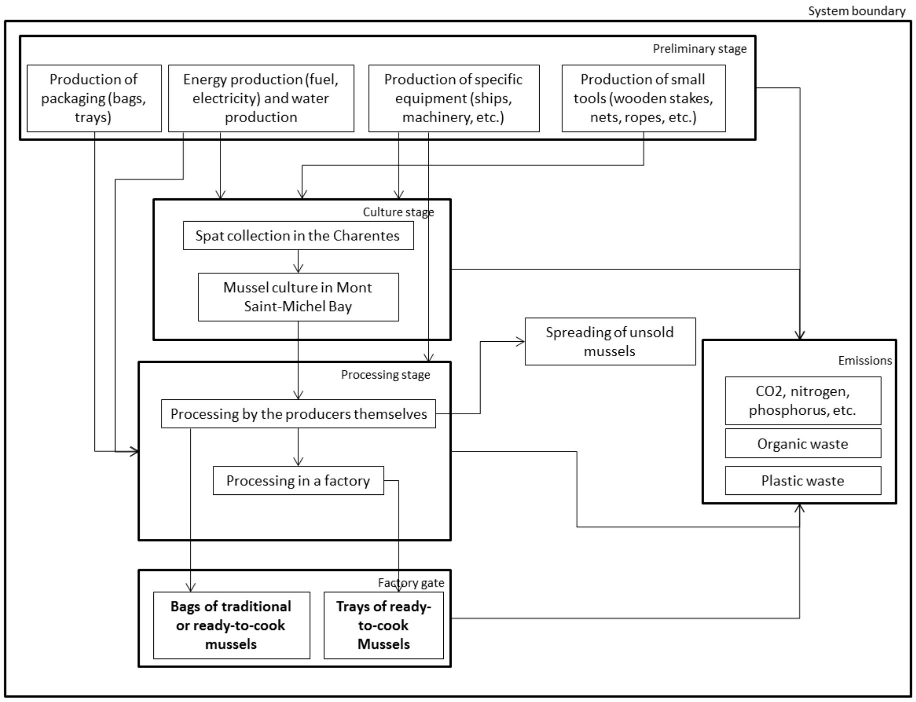

Figure 1. Stages of the mussel-production system included in the LCA

2.2. Life cycle inventory

Six of the main producers, operating in two or three subareas of the bay (Center, East, Far East), were interviewed from spring to summer 2012. They produced 300-450 tonnes of mussels per year. Their practices are regulated according to AOC specifications, which restrict the percentage of bouchots in use at the same time (55-65% according to the area) and the maximum mean annual yield to 60 kg of mussels per bouchot. These rules aim to preserve the productivity of the bay, and some producers have chosen to use only 50% of their bouchots at the same time. Bouchots are square wooden stakes, 5-6 m long and 10 cm wide, sunk 1 m deep in the foreshore. The stakes are made of exotic woods, most often *Lophira alata* and *Dinizia excelsa*, and are expected to last over 15 years. Bouchots are arranged in pairs of lines 100 m long and 25 m apart. The mussels must remain a minimum of 8 months on the bouchots. The beginning of harvest season occurs once the mussels have reached 4 centimeters in length and meet criteria of condition index and taste. This usually occurs in the first half of July. Mussels are harvested using hydraulic arms supported by amphibious boats, which scrape the whole mussel assemblage off, leaving the bouchots clean.

Water conditions in Mont Saint-Michel Bay are not suitable for natural reproduction of the blue mussel. Therefore, spat collection starts in March in the Bay of Biscay, mostly in the Charentes region. We described this stage using data provided by one producer, who is responsible for producing 300 km of seeded ropes each year. Producers construct temporary wooden structures on the lowest parts of chosen beaches and then extend ropes made from hemp or coconut fibers on these structures. The ropes remain in place for about two months before being brought to Mont Saint-Michel Bay by road.

Upon arrival, seeded ropes are coiled around free bouchots or laid out on temporary horizontal structures built on the foreshore until new bouchots are available. Once mussels start growing, producers stabilize the structure of the bouchot by progressively encasing each in 3-7 polypropylene nets, depending on the exposure of their sites. To reach the culture sites, producers use tractors, small aluminum boats or amphibious boats. Specific data collection on amphibious boats was performed at the main shipbuilding company.

After harvest, mussels are placed for at least 24 hours in an oxygenated purification pool. Water is pumped from the bay into a lagoon shared by all shellfish producers in the area. Afterwards, mussels go through a declumper, which separates the mussels from each other and shreds the ropes and nets. They are then cleaned in a brushing machine and calibrated, first mechanically, then by hand, to remove any other kind of waste (crabs, algae, broken mussels, etc.).

At this point, 20-30% of the production is discarded, mostly due to undersized mussels. These mussels are currently spread over the tractor roads in the bay. This practice serves two goals: stabilizing the roads in the sand and driving mussel predators away from the bouchots by giving them an easily available food source. Mussel spreading is currently under heavy debate and may soon be discontinued.

Mussels may then be put in 5, 10 or 15 kg polypropylene bags of "traditional" mussels and put up for sale. They may also pass through another machine that removes the byssus threads and be sold in 5, 10 or 15 kg polypropylene bags of "ready-to-cook" mussels, or 1 kg polypropylene trays of "ready-to-cook" mussels. LCA was performed on each type of packaging. However, this paper shows the LCA results for "ready-to-cook" mussel's bags only. Organic waste from cleaning and processing stages and all other types of waste are buried at the local landfill. Wastewater is returned to the ocean without any kind of treatment. All back ground data was extracted from the ecoinvent v2.2 database (Swiss Centre for Life Cycle Inventories 2010).

Mussel shells are composed of $CaCO_3$ and may be considered a carbon (C) sink if they are degraded slowly. According to Fry (2012) and considering that coastal seawater is saturated in $CaCO_3$ and that the Mont Saint-Michel Bay is shallow, we treated spread mussel shells as C sinks. Since other shells are expected to join ordinary household waste, we examined common waste-management options in France. Since 44.4% of French household waste ends up in landfills (ADEME 2008), we assumed that the shells of 44.4% of all mussels sold in France would be buried and act as a C sink.

Spread mussel flesh was excluded from our calculations since it belongs to a short cycle and did not influence overall nutrient balance of the bay. We calculated N and P exports based on the amount of mussel flesh present in marketed mussels. We also assumed increased biochemical oxygen demand of the water released into the bay due to organic-matter processing and bacterial decomposition of discarded mussels.

We estimated amounts of C, N and P in mussels (Table 1) based on Brigolin et al. (2009) and Chairattana et al. (2012). With data from the former, we excluded mussel respiration, considered as a short cycle in the bay, and calculated the amount of C sequestrated in mussel shells based on the balance of the remaining C flows. We supplemented this analysis with data from Chairattana et al. (2012), who showed that part of mussel shells is formed from dissolved CO_2, (not included in Brigolin et al.(2009)).

Table 1. Estimated C content in the shell, and N and P contents in the flesh, per tonne of blue mussels in the life cycle inventory.

Element in shell or flesh	Per tonne of harvested mussels
C in shell from ingested C (Brigolin et al. 2009)	198 kg C
C in shell from dissolved C (Chairattana et al. 2012)	18 kg C
N in flesh (Brigolin et al. 2009)	4.17 kg N
P in flesh (Brigolin et al. 2009)	0.38 kg P

2.3. Life cycle impact assessment

Impact categories were selected based on previous studies and guidelines in aquacultural LCA (Aubin et al. 2013; Pelletier et al. 2007): climate change (kg CO_2-eq), acidification (kg SO_2-eq) and eutrophication (kg PO_4-eq) were calculated using characterization factors of CML2 Baseline 2000 version 2.03 (Guinée et al. 2002). Energy use (MJ) was calculated according to the Total Cumulative Energy Demand (TCED) method, version 1.03 (Frischknecht et al. 2004). Water dependence (m^3) refers to freshwater use and was calculated according to Aubin et al. (2009). Calculations were performed with Simapro® 7.0 software.

2.4. Interpretation of results

Impacts of mussels at each production site for each producer were calculated independently (13 observations) and then aggregated. Variability in impacts was considered but not uncertainty due to foreground and background data. Contributions of different production stages of the system to the impacts were studied.

3. Results

Impacts of one tonne of ready-to-cook mussels varied according to production area and producer: acidification from 1.47-2.65 kg SO_2-eq, eutrophication from -1.69 to -0.85 kg PO_4-eq, climate change from -44.7 to 125.5 kg CO_2-eq, TCED from 7666-12,866 MJ, and water dependence from 92-98 m^3 (Table 2).

Table 2. Means and standard deviations (SD) of impacts calculated for 1 tonne of “ready-to-cook” mussels, depending on their location of culture within Mont Saint-Michel Bay.

	Center		**East**		**Far East**	
Impact	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
Acidification (kg SO_2-eq)	2.24	0.25	2.29	0.26	1.59	0.15
Eutrophication (kg PO_4-eq)	-1.09	0.27	-0.98	0.14	-1.24	0.32
Climate change (kg CO_2-eq)	37.49	70.37	77.45	64.68	-6.56	34.81
Total cumulative energy demand (MJ)	11,360	1132	11,237	1599	8285	1069
Water dependence (m^3)	96.9	18.4	96.7	18.7	94.0	18.2

On-site transport was the main contributor to acidification and climate change impacts and a major one to TCED (Fig. 2). The culture stage greatly decreased eutrophication and slightly decreased climate change. Wooden stakes decreased climate change as much as on-site transport increased it. Wooden stakes were also an important contributor to TCED. The cleaning and packing stage was the main contributor to water dependence and a major one to TCED and eutrophication. The spreading of discarded mussels contributed markedly to eutrophication but slightly decreased climate change.

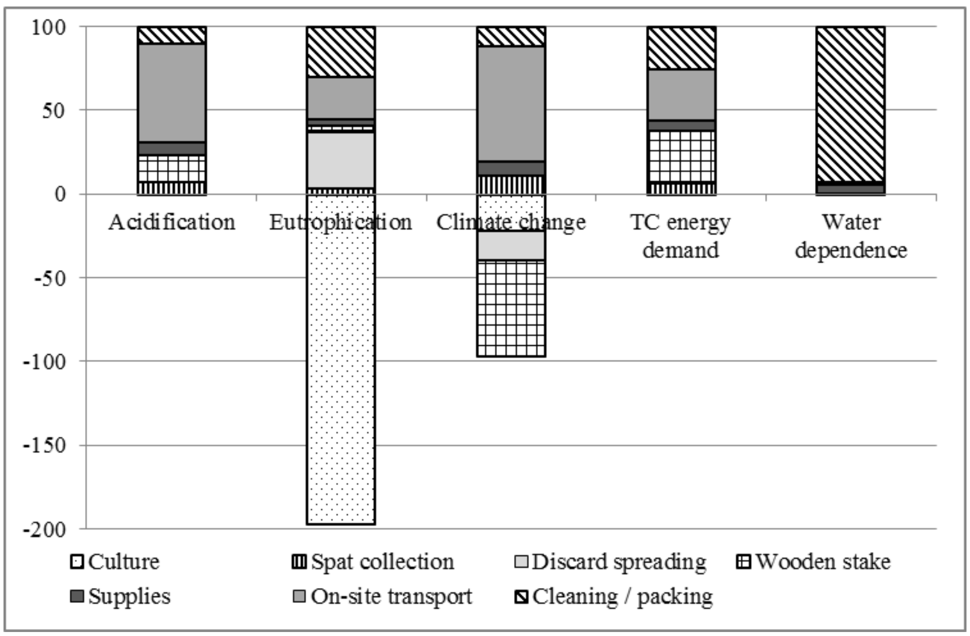

Figure 2. Contribution to environmental impacts of different stages of the production system of ready-to-cook mussels in Mont Saint-Michel Bay

4. Discussion

Assuming the uptake of N and P by mussel production induces a negative value for eutrophication impact, indicating a potential mitigating effect on water quality of the bay. Nevertheless, this favorable effect is low, around 1 kg PO_4-eq per tonne of “ready-to-cook” mussels, which should be compared to eutrophication impacts of other agricultural products produced in the Mont Saint-Michel region (e.g. 14 kg PO_4-eq per 1 tonne of pig live weight (ADEME 2013)). This result indicates a small but potential ability of mussel production to decrease eutrophication impacts due to livestock production at the territory level. This type of complementarity activity is also developed, for example, in integrated multi-trophic aquaculture for salmon production (Barrington et al. 2009).

Estimated climate change impact in this study showed high variability. On average, climate change impact is decreased first by the sequestered C in wooden stakes, then the spreading of discarded mussels, and then the burial of sequestered C in mussel shells in landfills. All of these processes, however, have high levels of uncertainty. Inventories from the ecoinvent database used to model exotic wood production are standard ones, and estimates of the wood's C sink effect are debatable. From this ecoinvent methodology, CO_2 assimilation is based on 49.4% of woods' C content. The percentage of household waste put into landfills may vary greatly depending on region and time period. Finally, the spreading of discarded mussels is about to be forbidden in Mont Saint-Michel Bay. In our case study, mitigation of climate change impact mainly compensates CO_2 emissions due to on-site transportation. At best, the bouchot blue mussel culture of Mont Saint-Michel Bay can be considered to produce little or no climate change impact but not as a C sink. Uncertainty analysis needs to be performed to assess the robustness of this conclusion.

In our study, impacts varied greatly according to the geographic location of bouchot sites. Two competing factors may explain this: mussel yield is higher in the Far East of the bay (due to ocean currents) than in the two other sites, but the Far East is further from on-shore infrastructure, and fuel consumption due to on-site transportation is a major contributor to impacts. As a consequence, we can observe negative values in eutrophication and climate change in the Far East area, where the production yields are the highest, inducing the highest nutrient uptake, and where the C fixing in shell over-compensate the CO_2 emissions of the rolling stock. This result has led the mussel producers' organization to revise their transportation strategy and the location of infrastructure.

5. Conclusion

Considering the chemical composition of mollusk flesh and shells to better describe their extractive capacity in the environment influences estimates of eutrophication and climate change impacts calculated with LCA. Results show a potential mitigating effect of blue mussel culture on water quality. A C sink effect due to mussel shell calcification was not confirmed, but a methodology that includes the composition and fate of shells during production processes and after consumption was developed. A similar approach should be performed on other production systems, other mollusks, and in other regions.

The bouchot culture of mussel is a low-impact activity, but fuel consumption to transport workers and mussels is a key point to analyze. Different environmental impacts in different locations suggest that efforts to render LCA increasingly spatially-explicit should be continued.

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Life cycle analysis of swine management practices

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ABSTRACT

A comparative Life Cycle Assessment (LCA) of different management strategies in the pork industry was performed to evaluate the impact on global warming potential (GWP) and energy and water use. The management strategies included immunocastration, production without ractopamine, production without antimicrobials, and use of gestation pens. Impacts of each management practice were compared to a common baseline production strategy that represents current production practices. Scope of this study was from cradle through farm gate with a functional unit of 1 kg of live weight at the farm gate. Each scenario was simulated using the Pork Production Environmental Footprint Model, to populate life cycle inventory inputs for SimaPro V7.3 (Pre' Consultants, The Netherlands). The results showed that compared to the baseline, use of gestation pens and production of immunocastrated pigs had lower GWP, energy use and water use. Production without ractopamine and antimicrobials increased GWP, energy use, and water use.

Keywords: Swine management practices, Life cycle assessment,

1. Introduction

There is a demand for changes to some of the management practices prevalent in the swine industry Florida, in 2002, banned use of sow gestation crates and similar ban was imposed on the swine industry in Arizona in 2006 (Mench 2008). A survey conducted by the Rutgers University in 2003 revealed that between 74 and 83% of the participants disagreed with practices such as tail docking of cows and pigs without analgesics and confining gestating sows (Mench 2008). Heeding its consumers, (Smithfield Foods) decided in 2007 to phase out gestation stalls on company-owned farms over next 10 years and replace them with pens.

Life Cycle Assessment (LCA) is an effective tool for performing trade-offs associated with alternate management strategies. This project supports the goal of the National Pork Board environment committee to: optimize management practices to enable producers to make informed management practice decisions to continually improve their farms; provide pork producers with the information and education they need to evaluate and implement appropriate management practices on their farms; and educate customers about the environmental and sustainability consequences of their purchase decisions.

The objective of this study was to quantify differences or establish the absence of differences in the global warming potential (GWP), cumulative energy demand, and water consumption between current practices and proposed alternates. All the scenarios were simulated for Wright County, Iowa and a typical meteorological year was used as the simulated climate.

2. Methods

This study analyzed the cradle to farm gate impacts for 5 different management practices (Table 1) with a functional unit of 1 kg live weight at the farm gate was chosen for this study.

Table 1. Proposed alternate management practices

Management strategy	Description
Immunocastration	Use of inmmune-castration methods/product(s)- Improvest®
No ractopamine	Removal of ractopamine (RAC) as a tool to improve growth and production
No GP antimicrobials	Removal of antimicrobials as growth promoters (GP)
No Prev. antimicrobials	Removal of antimicrobials to prevent emergence of herd infection in addition to removal of GP antimicrobials
Pen gestation	Use of pen gestation housing

2.1. Description of Models

Assessment and comparison of different management practices in terms of greenhouse gas emissions, water use and electricity use involved a two-step process. In the first step nursery, grow-finish and sow barns were simulated using Pig Environmental Footprint model (University of Arkansas). The outputs of these simulations were used in the second step for LCA analysis using SimaPro V7.3 (Pre' Consultant, the Netherlands).

2.2. Pig Environmental Footprint model

The growth and feed conversion performance of pigs, resource consumption, and emissions to the environment were simulated using the PPEF model. The model simulates pig growth, feed intake and water consumption, electricity and natural gas use, manure handling, and greenhouse gas emissions over an annual cycle. The PPEF model uses a growth performance model developed by the National Resources Council (NRC) to predict growth and feed consumption of pigs (National Research Council 2012).

2.3. SimaPro LCA model

The SimaPro software platform was used to compare each management scenario to the baseline and for uncertainty analysis. The Ecoinvent database, corrected for US electricity, was used for background processes, and feed production data for the US developed at UA was used for constructing the animal's rations.

2.4. Scenario Description

Each test scenario evaluated only one management practice, where only the key element was changed. A single, typical ration was used for all scenarios to prevent confounding effects of the ration with the practice under evaluation. The last dietary phase in grow-finish barn was formulated to accommodate use of ractopamine by adding 0.05% Paylean 9 (Rikard-Bell et al. 2009, See et al. 2004). It was assumed that pigs have ad libitum access to the water and drinking water consumption, along with cooling and wash water, were simulated in the PPEF model.

2.4.1. Baseline scenario

Each scenario was compared pairwise with the baseline scenario. The baseline scenario included male (barrows) and female pigs in equal numbers and assumed growth promoting antimicrobial (AGP) use in the nursery, preventive antimicrobial use as required, ractopamine use in grow-finish barn, with tail docking and surgical castration of male pigs performed in the lactation barn. The NRC growth model for growing pigs assumes that the maximum protein deposition value (Pdmax) in pig decreases after a certain weight is reached (National Research Council 2012). Paylean-9 was added to the diet in the last phase of feed formulation and at a pig body weight of 96 kg to simulate 28 days on the ractopamine. The average market weight of 125 kg (approx. 275 lbs) (National Pork Board , USDA , USEPA) was chosen for fair study.

2.4.2. Removal of ractopamine (RAC)

Ractopamine is a dietary supplement which improves average daily gain (ADG) and feed efficiency (FE) in finishing pigs (Armstrong et al. 2004, Barbosa et al. 2012, Dunshea et al. 1993, Hinson et al. 2011) and is usually added to the diet during last 28 days in the finishing phase (Hostetler et al. 2012). The final ration phase, without ractopamine, was also altered to include more corn based on recommendations from swine nutritionists. No changes were made to the nursery and sow barn that provided feeder pigs to the grow barn for this scenario.

2.4.3. Antimicrobial use

Antimicrobials are used in animal industry for disease prevention, animal health improvement (Romina Ross et al. 2010), and as growth stimulants (Kiarie et al. 2011). However, antimicrobial use in animal industry has come under scrutiny due to the concerns about development of antimicrobial resistant strains that could affect human health (Holt et al. 2011). We constructed scenarios to evaluate the impacts associated with reduced use of antimicrobials in pig production. The first scenario assessed impact of eliminating growth promoting antimicrobials (AGP), while the second scenario assessed impacts of eliminating both growth promoting and preventive antimicrobial (NoPrev) use.

2.4.4. No growth promoting antimicrobials

Williams et al. (1997) estimated 12.7% increase in the metabolic maintenance energy requirement (MMER) when pigs have poor health. To simulate elimination of AGP from the production in the nursery phase, the MMER in the NRC growth equations for pigs between 5 and 23 kg was increased by 12.7%. The grow-finish phase of the production was assumed to be unaffected by elimination of AGP, as far as pig performance is concerned.

The National Pork Board Taskforce suggested not using AGP could mean fewer pigs would reach the expected weight and size requirements in the production facility, which was estimated to increase voluntary cull rate in the nursery and grow-finish barn to 0.25% for median health facilities. Without AGP, the mortality rate was expected to increase by 0.2% in the nursery phase. Because AGP is used mostly in the nursery phase, production without use of AGP was expected to have no impact on mortality rates in grow-finish barn.

2.4.5. No growth promoting and preventive antimicrobials

In this scenario, the effects of production without use of either AGP or preventive antimicrobial use on performance parameters were estimated. When herd health is trending downward, antimicrobials are used prophylactically to reduce the chance of herd-wide infection. Animals which become sick are treated therapeutically and will recover or die. Without preventive use, more animals are likely to need therapeutic doses. Whittemore et al. (2001) reported that chronic diseases in pigs increase the maintenance energy requirements by up to 1.3 times the normal predicted value. However, in the current scenario it was assumed that not using preventive antimicrobials in the grow-finish barn does not necessarily mean the pigs fall sick. It was assumed that without AGP and preventive antimicrobials in the production, the performance of pigs would be poor compared to the baseline. Therefore, MMER of pigs in the nursery and grow-finish barns was increased by 12.7% (Williams et al. 1997) and 15% respectively.

Without AGP and preventive antimicrobial use in the production the voluntary cull rates and mortality was expected to increase by 4% in the nursery and 5 and 5.5% respectively in grow-finish barn.

2.4.6. Immunocastration

Surgical castration of male pigs without anesthesia within first 1 to 2 weeks of age is a standard industry practice (FAO , Thun et al. 2006). Besides preventing boar taint in the meat, which is a result of skatole levels higher than 0.2 $\mu g\ g^{-1}$ of fat, surgical castration also improves meat quality and suppresses aggressive behavior in pigs (Dunshea et al. 2001, Morales et al. 2010, Thun et al. 2006). However, surgical castration is under scrutiny of animal welfare groups because the procedure is painful and distresses pigs (Millet et al. 2011, Morales et al. 2010).

An alternative to the surgical castration that is being studied is immunocastration, which involves administering male pigs a dose of gonadotropin-releasing hormone (GnRH) which creates antibodies against GnRH and reduce skatole production. Dunshea et al. (2001) reported that immunocastration also reduces size of testes in male pigs suppressing sexual aggressive behavior. This compound is administered at about 9 weeks of age and then again between 3 and 10 weeks prior to slaughter. Because male pigs perform like boars, which means they gain leaner muscle mass, until the second dose of GnRH compound, immunocastration offers improved ADG and FE in male pigs compared to surgically castrated pigs (Batorek et al. 2012).

The performance of immunocastrated (IC) pigs was modeled as a split sex barn with half males and half gilts. The NRC growth model for growing-finishing pigs assumes not effect of pig sex on the MMER (National Research Council 2012). However, it is assumed in the model that entire males have lower metabolizable energy intake (MEI) compared to the barrows.

Effects of immunocastration after second injection were captured in the model by increasing estimated MEI by 21% and reducing MMER and Pd by 12 and 8% respectively. No changes to the diet formulation were made for the immunocastration scenario.

2.4.7. Gestation stalls

Gestation stalls offer benefits such as maximum barn space utilization and controlled feeding, but the management practice has criticized by animal welfare groups because the stalls offer sow minimum or no free movement (Lammers et al. 2007). Group housing reduces the stocking density and thus requires additional housing to maintain animal production. Due to the difference in barn infrastructure necessary for the alternate management using gestation pens, the LCA scenarios have included the effect of changes in the infrastructure. A 10 year life for the barn facility, including the stalls and pens was assumed for this scenario. A bill of materials for construction of sow, nursery and grow-finish barns from plans published by Iowa State University Midwest Plan Service was created (data not published).

This scenario was designed to evaluate environmental impact of production management using gestation pens. Data for comparison between gestation stalls and group pen housing were obtained from published articles. This scenario evaluated the option of using gestation stalls for the entire gestation period only. It was assumed that farrowing stalls were used for both group pen and individual stall scenarios. The differences between sows housed in gestation stalls and in group pens were observed in number live births, litter size, pre-weaning mortality, and piglet weights at birth. An analysis of data obtained from peer-reviewed articles was performed to prepare scenarios for group and gestation housing (Table 2). These numbers were close to the results reported by (McGlone et al. 2004).

Bates et al., (2003) reported that 72% of sows housed in group returned to estrus within 7 days compared to 68.4% of sows in gestation stalls. In addition, 94.3% of group housed sows remained pregnant after initial service compared to 89.4% of sows in stalls. These differences were capture in the PPEF model by adjusting the average number of days between piglet weaning and insemination.

Table 2. Production parameters for gestation stalls and group housing

	Gestation stalls	**Group pens**
Litter size	10.5	10.34
No born alive	9.55	9.41
Preweaning mortality	15.4%	16.3%
Pigs weaned	8.08	7.88
Weight per piglet	1.5	1.53
Backfat thickness (mm)	19.6	20

Averages of data obtained from Anil et al. (2005), Bates et al. (2003), Harris et al. (2006), Lammers et al. (2007), SCHMIDT et al. (1985), Wang et al. (2011)

3. Results and discussion

Results of the LCA for the baseline and five pork production management strategies are presented in Analysis of the changes in environmental impact category metrics as a result of changed inventory from the PPEF for each management strategy showed that some strategies increased impacts, while others decreased impacts (Figure 1, 2 and 3). These analyses represent simulated estimates of impacts and should be interpreted as potential trends rather than absolute estimators.

3.1. Ractopamine

Not using ractopamine resulted in 6.5% increase in global warming potential, 4.6% increase in fossil fuel increase, and 5.6% increase in water use. The driving factor for the increase in GWP from removal of ractopamine was lowered productivity during the last month of finishing. The model simulations showed that days in the barn are increased when RAC is not used in the production. This directly affects the quantity of feed consumed, manure produced, and requires a small increase in necessary barn infrastructure to support the same annual lean pork production (i.e., a decrease in the number of turns per barn per year).

3.2. Antimicrobials

Not using antimicrobials as a growth promoting strategy resulted in 1.6% increase in global warming potential, 1.8% increase in energy use, and 3% increase in water use. The increased GWP was driven by two factors: lowered daily gain and feed efficiency leading to increased feed consumption and time required to reach market weight, and thus additional manure production as well. If preventive use of antimicrobials is avoided, there will be additional mortality, as well as further decrease in daily gain and feed efficiency compounding the effect. Finally, additional barn infrastructure will be needed due to lengthened time to reach market weight.

Coupling removal of AGP with not using antimicrobials for disease prevention resulted in 17.4% increase in global warming potential, 18.6% increase in energy use, and 18.9% increase in water use, the largest changes in this assessment. The effects of not using antimicrobials for disease prevention were driven by the same process impacts as not using them for growth promotion, compounded by increased mortality across the entire herd.

3.3. Immunocastration

This management alternative resulted in 2.3% decrease in global warming potential, 2.5% decrease in energy use, and 1.9% decrease in water use. This alternative approach to controlling boar taint resulted in increased average daily gain and reduced daily feed intake compared to baseline. In the grow barn, using immunocastration also resulted in increased cycles per year. This resulted in less overall feed consumption and manure production as well as a small reduction in the necessary barn infrastructure associated with the faster average turn-around for the barns.

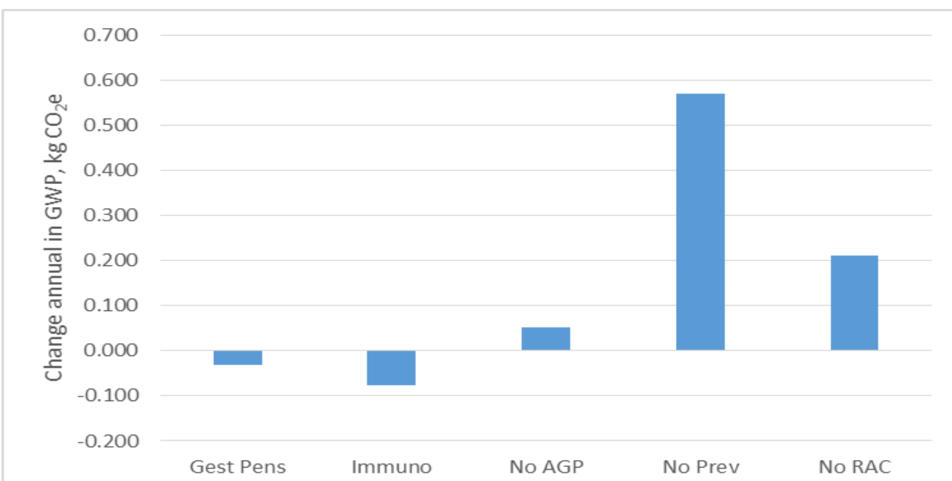

Figure 1. Estimated potential change in annual global warming potential (kg CO_2e) resulting from US production strategies for 1 kg live weight at farm gate

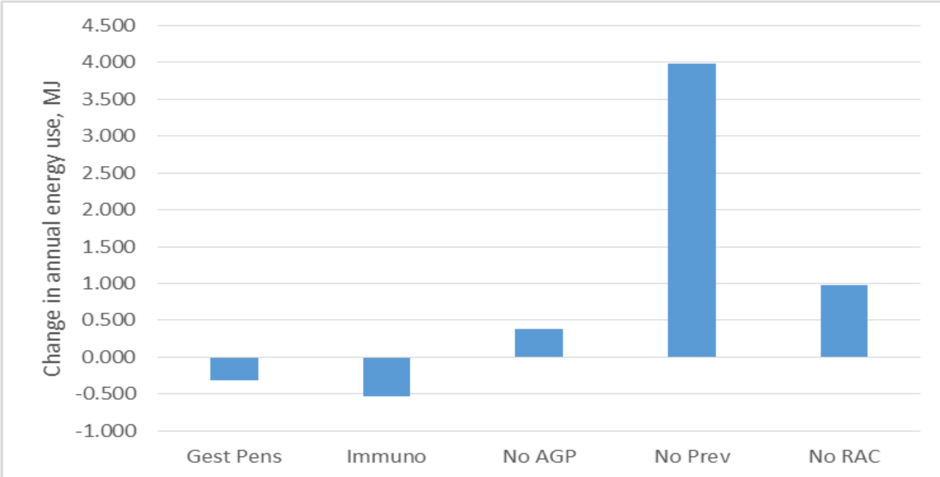

Figure 2. Estimated potential change in annual energy use (MJ) resulting from US production strategies for 1 kg live weight at farm gate

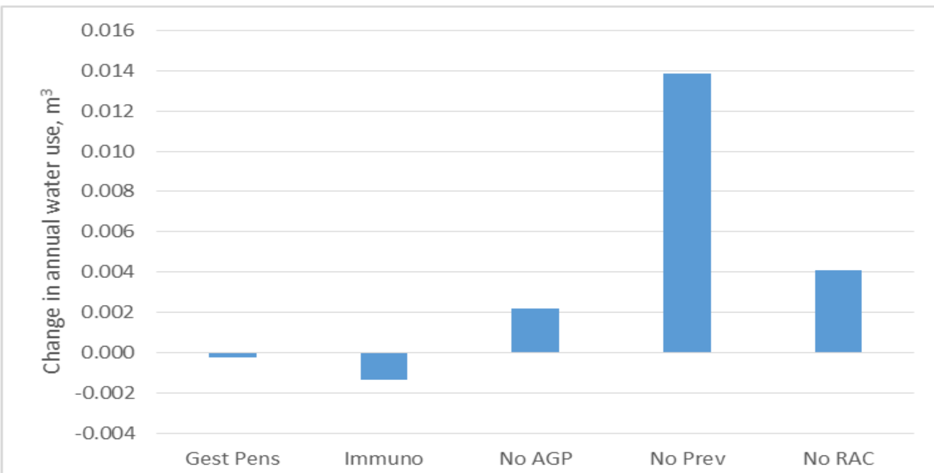

Figure 3. Estimated potential change in annual water use (m^3) resulting from US production strategies for 1 kg live weight at farm gate

3.4. Gestation pens

Using pen gestation structures rather than stall gestation structures resulted in 1% decrease in global warming potential, 1.5% decrease in energy use and 0.3% decrease in water use. Lower GWP observed in this scenario was a result of lower feed consumption and lower manure emissions. However, the barn infrastructure requirements for pens are 65% larger, based on our modeling of the space requirements for sow in stalls compared to pens. This increase the GWP, which is amortized over the expected life of the barn, and essentially offsets the lower GWP observed for this scenario. The lower energy demand appears to be a result of lower electricity use for fans observed for gestation pens.

4. Conclusion

Life cycle analyses of five pork production strategies for three environmental impact categories for yielded a range of results, from 17% increase in global warming potential (removing AGP and preventive antimicrobials) to approximately 2.5% reduction in energy use (immunocastration). Based on LCA results, the following pork production strategies increased environmental impact metrics across all three impact categories: not using antimicrobials for growth promotion or disease prevention and not using ractopamine for growth promotion. Conversely, using immunocastration and pen gestation production strategies decreased global warming potential and energy use, and water use. The results of this study indicate that changes made to production practices in the swine industry, could affect sustainability metrics and therefore need cautious evaluation. These results however, are the product of simulation of pork production strategies combined with unit process LCAs; these models are very sensitive to time in the barn at each growth stage, temperature inside the barn, and mortality rates and therefore should be interpreted cautiously.

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Environmental impacts of imported versus locally-grown fruits for the French market as part of the AGRIBALYSE® program

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ABSTRACT

As part of the AGRIBALYSE® project, apple and peach from France, small citrus from Morocco and mango from Brazil were evaluated. Representative systems for each fruit were designed relying mostly on expert knowledge for apple, peach and small citrus and on a detailed survey of 8 commercial orchards for mango from the Rio San Francisco Valley in Brazil. For most impact categories, apple showed the least impacts, small citrus the greatest. Beyond the classical yield effect, this was mostly linked to lower fertilizer rates for apple and to fossil energy share in the electricity. For marine eutrophication, mango and small-citrus had the least impact, followed by apple and peach far above. For ecotoxicity, mango had the least impact followed by apple, peach and small citrus far above. Ecotoxicity results revealed the most uncertain due to the difficulty to determine representative crop protection practices for perennial crops. Complementary research is needed to better model crop protection practices, field emissions and water use impacts.

Keywords: Fruits; environmental impacts; crop protection practices; uncertainty; representative systems

1. Introduction

As all other food products traded globally, fruits are under growing scrutiny regarding their environmental impacts. In France, the Agribalyse® program was launched by the French environment agency (ADEME) in 2009 to support environmental labeling as planned by the "Grenelle de l'environnement" roundtables. Based on the life cycle assessment (LCA) methodology, the objective of Agribalyse® was the development of a homogenous and consensual LCI database for French agricultural products and a few imported products. In France, 50% of fruits are produced locally and 50% are imported often from distant and warm countries. As part of the Agribalyse® program, two locally-produced fruits: apple and peach, and two imported from overseas: small citrus from Morocco and mango from Brazil were evaluated. The evaluation of fruits with LCA is quite recent, the most studied fruits being citrus (Sanjuan et al. 2005; Beccali et al. 2009; Trydeman Knudsen et al. 2011; Pergola et al. 2013; Lo Giudice et al. 2013) and apple (Mila i Canals et al. 2006; Mouron et al. 2006; Alaphilippe et al. 2013). Ingwersen (2012) recently published a full LCA study on pineapple from Costa-Rica, but tropical fruits have been seldom studied with complete LCA studies. The application of LCA to fruit cropping systems has revealed specific challenges related mostly to their variable and perennial cropping systems, frequent pesticide treatments and use of irrigation water (Mouron et al. 2006; Bessou et al., 2012; Cerutti et al., 2011 and 2013). Although most LCA studies do not account for the perennial cycle of fruit cropping systems, certain authors have recommended the inclusion of all phases of fruit orchards in the LCA modeling of fruits including nursery, orchard installation, growing of trees, full production phase and possibly decreasing-yield phase and dismantling of plantation (Mila i Canals et al. 2006; Cerutti et al. 2011; Bessou et al. 2012; Cerutti et al. 2013). Bessou et al. (2012) proposed a formalization of the different possibilities to account for the perennial cropping systems depending on the objective of the study and data availability. One can either use a spatial assessment, a chronological assessment or a modular assessment, presented as the minimum requirement to account for the perennial cropping cycle. A modular assessment in which each phase is modeled independently with different sources of data can be used when neither complete spatial data nor chronological data are available on the studied system. Bessou et al. (2012) also highlighted the inadequacy of usual methods for estimating field emissions for perennial cropping systems especially under tropical, sub-tropical or semi-arid conditions. For instance, tropical and sub-tropical systems remain clearly underrepresented in IPCC Tier 1 data sets. Bessou et al. (2012) concluded on the need for producing specific data sets on perennial cropping systems to improve existing operational models and the prediction of their field emissions. Finally, several authors also raised the issue of the choice of functional unit and allocation procedures for fruit products, insisting on the various qualities of fruits including their edible content. Ingwersen (2012) used the serving of fruit (165 g fresh fruit according to USDA 2009 definition) to express his results and compare with other LCA studies. Cerutti et al. (2013) recommended indicating the edible content of fruits when a mass-based functional unit is chosen. Regarding the comparison of imported fruits with locally-grown equivalent, the question of allocation may be crucial since the fruits exported correspond to the highest quality fruits, the lower quality fruits being usually sold locally.

In a context of recent application of LCA to the fruit sector, the objective of the Agribalyse® program was not to develop new research but to properly apply the consensual and up-to-date methodology for all agricultural products including fruits.

The objectives of this paper are:

- To present the methods and data used to design and assess fruit cropping systems in each situation
- To present and discuss the LCA results for the 4 fruits in relation to existing literature, methodological choices, data availability and studied function
- To identify some margins of improvement and research perspectives

2. Materials and methods

2.1. Goal and scope

The main objectives of this study were:

- Comparing a panel of major fruits produced with conventional rules and consumed in France, two locally-grown: apple and peach and two imported: mango from Brazil and small citrus from Morocco.
- Applying the consensual methodology of the AGRIBALYSE® project (Koch and Salou, 2013).
- Designing most representative systems as possible for each situation given the data availability.

In line with the Agribalyse® method, the functional unit used was 1 kg of fruit at farm-gate. Representative systems for each fruit were designed relying mostly on expert knowledge for apple, peach and small citrus and on a detailed survey of 8 commercial orchards for mango from the Rio San Francisco Valley in Brazil. The reference period defined in the Agribalyse® report is 5 years from 2005 to 2009 but must reach 10 years for strongly alternating productions such as fruits. This was formally possible for mango where data were collected over more than 20 years on real orchards but relied on expert advice for other fruits supposed to include seasonal and regional variability over the orchards' life. For crop protection practices, data were based on a large sample survey (349 field surveys) for apple, on expert advice for peach and small citrus and on average data for the 8 surveyed orchards for mango. For all fruits, the full orchards' life was modeled according to recent practice (2000-2010) as presented in section 2.2.1 using either real data or expert advice.

2.2. Studied systems

In agreement with the AGRIBALYSE® method, the system boundaries were set from cradle-to-farm-gate including the production, transport and use on the farm of all inputs except very minor tools and inputs, e.g. pruners, and non-agricultural buildings (Figure 1).

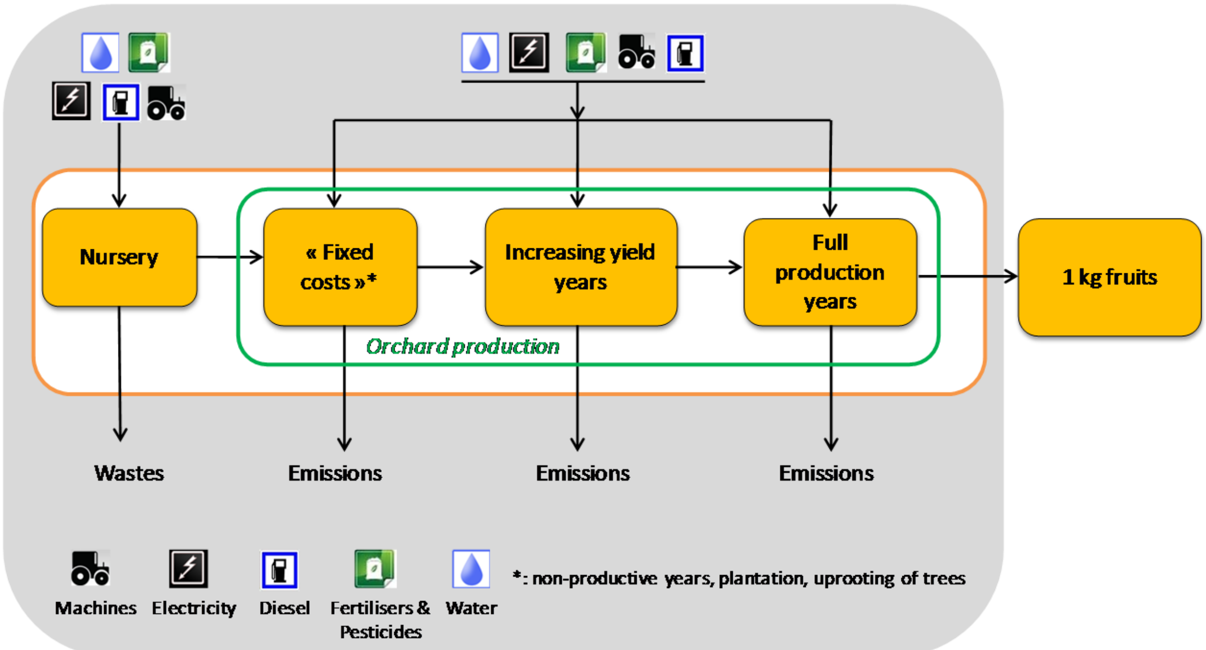

Figure 1. Cradle-to-farm-gate fruit systems for apple and peach from France, mango from Brazil and small citrus from Morocco

2.2.1. Modelling of perennial cropping systems

According to the AGRIBALYSE® methodology, the perennial cropping system was modeled in four phases: nursery (or plant production), fixed costs (including plantation, non-productive years and uprooting of trees at the end of the orchard life), increasing yield years and full production years. This implied to collect or estimate representative data for all four phases in terms of agronomic practices and duration. An important assumption for perennial cropping systems is the life time of the orchard. It was assumed to be 20 years for apple, 15 for peach, and 25 years for mango and small citrus.

2.2.2. Apple from France

Apple is mainly produced in the South-East, the South-West and the Loire Valley regions in France, according to 38%, 31% and 23% surface-wise, respectively (Agreste, 2008). In each region, experts of apple production from technical extension services and farmers' associations participated in the design of most representative apple production systems for the recent period. The average conventional system was a combination of non-scab and scab resistant (or tolerant) varieties across all regions weighted by their respective share. Crop protection practices were based on a survey from a large sample of orchards (349 field surveys) for the period 2005-2008. These data for the full production phase (years 5 to 20) were extrapolated to the entire life cycle of the orchard ignoring the constant evolution of active molecules certification. For the first (non-productive) years, crop protection practices were assumed to be one third of that for full production years while they were assumed to be two thirds for increasing yield years (from 2 to 4). Only the most common molecules were selected from the survey. Data for the nursery phase was based on the survey of two nurseries, one in the Loire Valley region the other in the South-East region. Key agronomic data for the full production phase of apple production and other fruits are presented in Table 1.

2.2.3. Peach from France

The production of peach is mainly located in the South of France. Similarly to the apple inventory, experts of peach production from technical extension services and farmers' associations were involved in the design of the most representative peach production systems for France for the recent period. Based on national statistics from Agreste (2008), the national average system was the weighted (surface-wise) combination of early, median and late productions, influencing the yield, mechanization requirements, crop protection and irrigation practices (see table 1 for full production phase). Two commercial nurseries representing more than 25% of the production of peach scions and grafted plants were surveyed to design the nursery phase. Crop protection practices were based on expert knowledge for the full production phase (years 5 to 15) and extrapolated to other phases of the orchard assuming one third of pesticide inputs from the full production was applied for the first non-productive years (years 1 and 2) and two thirds for increasing yield years (years 3 and 4).

Table 1. Main agronomic data for the full production phase of apple (France), peach (France), mango (Brazil) and small citrus (Morocco). Values are given per annum.

Intervention	Unit	Apple	Peach	Mango	Small citrus
Country		France	France	Brazil	Morocco
Orchard age	Years	20	15	25	25
Density	Trees/ha	1730	640	280	500
Yield	t/ha	53,7	28	33	28
Fertilisation					
N	kg/ha	50	110	165	180
P_2O_5	kg/ha	30	100	100	45
K_2O	kg/ha	125	220	273	180
Irrigation					
Water	m^3/ha	2767	7000	7999	8000
Energy	MJ/ha	2,988	7,560	2,946	22,830
Plant protection products					
Total herbicides	kg/ha	3.4	4.4	0	10.2
Total insecticides	kg/ha	5.1	0.7	0.301	9.58
Total fongicides	kg/ha	38.4	24.2	5.66	16.5
TOTAL pesticides	kg/ha	48.9	29.3	5.961	36.3
Growth regulators	kg/ha	0.2	0	4.03	0.02
Petroleum oils	kg/ha	12.3	16.3	0	0

2.2.4. Mango from Brazil

Brazil is the leading supplier of fresh mangoes to the EU. In the Rio San Francisco Valley which concentrates more than 90% of Brazilian mango exports, modern and intensive production systems have developed. These systems feature year-round production thanks to well-controlled floral induction and abundant dam water access. In this region, a sample of eight contrasted Kent and Tommy Atkins mango orchards was surveyed in 2012. Data over the complete crop cycle of mango trees was collected, over more than 20 years for elder orchards. Despite this detailed and very time consuming survey, many input and yield data were missing over the 25 years of mango orchards. Annual average for all input and yield data available across the eight orchards were first calculated and aggregated into average data for each phase (see table 1 for full production phase). No nursery was included since grafted plants are produced on farm.

2.2.5. Small citrus from Morocco

In Morocco, small citrus for export to France are produced in two main regions of production: the Souss region and the Oriental region (Berkane area) with, for the 2009-2010 season, 55.6% of small citrus exported and 32.8%, respectively (EACCE, 2010). Until recently in each region, specific varieties and cropping system management were used. In the Oriental region, traditional practices included mainly Clementine varieties such as Cadoux, low density orchards (270 trees.ha^{-1}) and gravity irrigation, while in the Souss region the management was more modern and intensive using mostly the Nour variety, high density orchards (500 trees.ha^{-1}) and drip irrigation. According to local experts, the Oriental system is rapidly evolving toward a more modern management very similar to the Souss system. For this reason, we chose the Souss system using Nour variety, high density orchards and drip irrigation as the most representative for the Moroccan small citrus for export to France.

Key input and yield data for the representative Souss-Nour system were based on expert knowledge of the small citrus production in Morocco for each phase of the citrus orchard (see table 1 for full production phase). Other more specific operations and data were based on a detailed survey over the whole orchards' life of one commercial orchard of small citrus from the Beni Mellal region. Regarding crop protection practices, the main pests were inventoried and the most common practices and active molecules used for each pest defined. Other practices may exist. A detailed survey was conducted in a commercial nursery to design the nursery phase.

2.3. Environmental inventory

2.3.1. Emissions from orchards

To estimate field emissions, the AGRIBALYSE® recommendations were applied (Koch and Salou, 2013). Phosphates and pesticides emissions were calculated according to Nemecek and Kägi (2007), assuming that 100% of the pesticides applied would be emitted to the soil (Nemecek and Kägi 2007). Nitrous oxide, carbon dioxide from urea and lime and nitrate leaching were estimated according to IPCC (2006). Ammonia emissions were based on emission factors from EMEP/CORINAIR 2006 and nitrogen oxides according to EMEP/EEA (2009). According to IPCC (2006), nitrate leaching was considered nil for mango and small citrus because localized irrigation is used and rainfall is reduced in both regions (daily irrigation (or rainfall) volume was constantly below the soil field capacity) while for apple and peach it was assumed to be 30% of the nitrogen inputs. The SALCA-SM method was used for trace elements but only for French products since data was missing for imported fruits (Freiermuth, 2006 and SOGREAH, 2007). For land transformation, the Ecoinvent v2 reference was used (Frischknecht et al., 2007).

2.3.2. Indirect inventory data

Indirect inventory data were based on Ecoinvent Life Cycle Inventories (LCI) database available in the SIMAPRO software and on processes developed specifically for the studied production system and the country.

2.4. Characterization of environmental impacts

The impact assessment was performed using the ReCiPe Midpoint life cycle impact assessment method (Goedkoop et al., 2009), adopting the Hierarchist perspective. The following environmental impact categories were considered: climate change (100 years; kg CO_2eq), terrestrial acidification (g SO_2eq), freshwater and marine eutrophication (g P-eq and g N-eq respectively, based on the nutrient-limiting factor of the aquatic environment), terrestrial and freshwater ecotoxicity (g 1,4-DB-eq: 1,4-dichlorobenzene), agricultural land occupation (m^2.year), fossil depletion (kg oil-eq). The non-renewable energy consumption (fossil and nuclear; MJ-eq) was assessed using the Cumulative Energy Demand method (Hischier et al. 2009). To facilitate comparison with published LCA studies, we also calculated LCIA results using the CML 2001 methodology (Guinée et al., 2002) (see section 2.5).

2.5. Comparison with published LCA studies

We compared our cradle-to-farm-gate LCA results with cradle-to-farm-gate LCA results from 9 published studies on fruits using the CML 2001 methodology (Guinée et al., 2002) (Table 2). Incomplete LCA studies were discarded. Among all studies, GWP and Non-renewable energy demand were the most consistently evaluated and could be systematically reported. Apart from Pergola et al (2013) who evaluated both non-renewable and renewable energy sources based on Namdari et al. (2011), all other authors used different versions of the Ecoinvent method for cumulative energy demand in MJ (Frishknecht et al., 1996; Frishknecht et al., 2003; Hischier et al., 2009). For eutrophication and acidification potentials, most studies used CML 2001 or EDIP97 which are identical for eutrophication and slightly different for acidification (Dreyer et al., 2003). Several studies did not include toxicity impacts due to methodological limitations (Beccali et al., 2009; Trydeman Knudsen et al., 2011; Pergola et al., 2013). In other studies a range of approaches was used for toxicity impacts. We only selected results from studies using the CML methodology: Sanjuan et al. (2005), Alaphilippe et al. (2013).

3. Results

3.1. Cradle-to-farm-gate LCA results for apple, peach, mango and small citrus

Except for marine eutrophication, terrestrial ecotoxicity and freshwater ecotoxicity, apple revealed the least impacting per kg of raw fruit at-farm-gate, followed by mango, peach and small citrus showing the greatest impacts (from twice to four times apple's impacts) (Table 2; Figure 2). This was firstly due to the yield of raw fruits ranging from 54 t/ha at full production for apple, to 33 t/ha for mango and 28 t/ha for peach and small citrus. A second important aspect was the fertilizer rates on orchards increasing from apple, peach, mango and small citrus. Overall, the two imported fruits showed higher fertilizer rates than the French ones. Another reason for this ranking was the share of fossil energy in the mix electric in each country, increasing from France (less than 10%), Brazil (10%) and Morocco (50%).

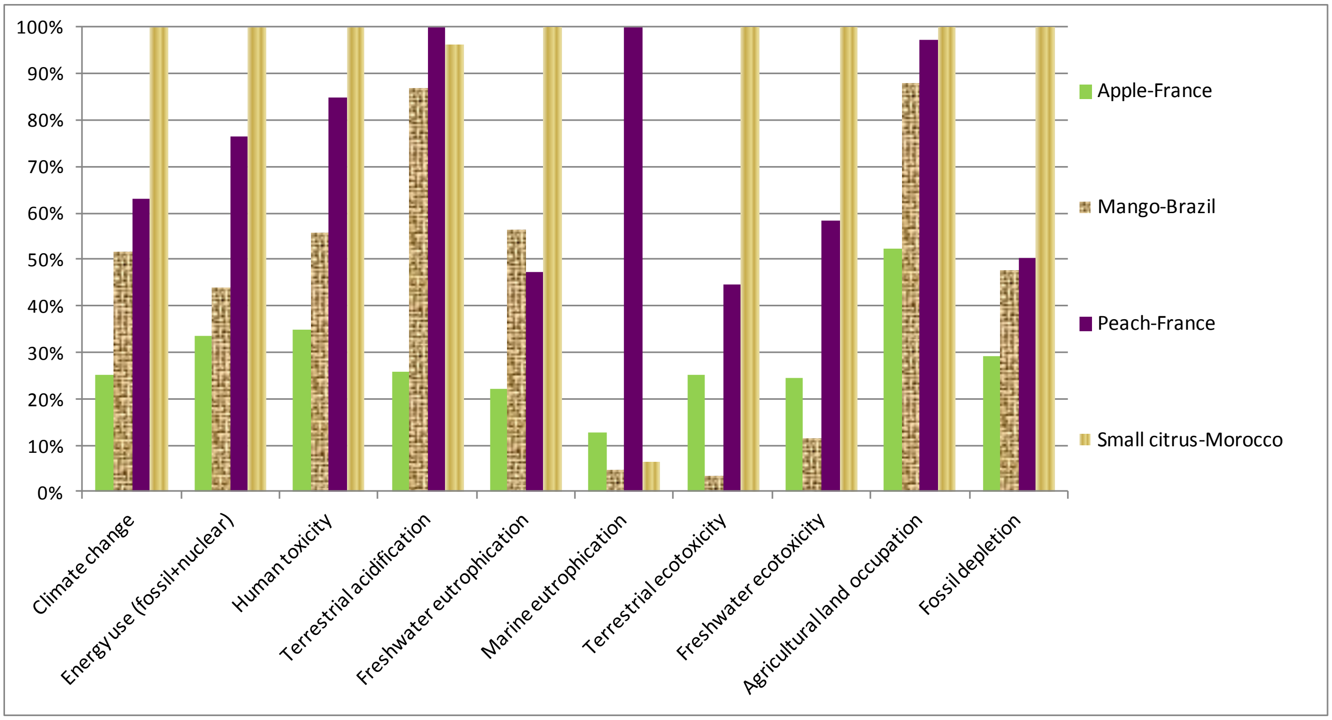

Figure 2. Cradle-to-farm-gate LCA results per kg of raw fruit for a selection of environmental indicators (ReCiPe Midpoint (H)) for apple, mango, peach and small citrus. Results are expressed as a percentage of the greatest result.

Table 2. Cradle-to-farm-gate LCA results per kg of raw fruit for a selection of environmental indicators (ReCiPe Midpoint (H); Cumulative Energy Demand) for apple, mango, peach and small citrus.

	Apple France	Mango Brazil	Peach France	Small citrus Morocco
Climate change (kg CO_2-eq)	0.0678	0.139	0.170	0.269
Non-renewable energy (MJ)	1.12	1.46	2.54	3.32
Human toxicity (kg 1,4 DB-eq)	0.0273	0.0436	0.0664	0.0783
Terrestrial acidification (g SO_2-eq)	0.610	2.05	2.36	2.27
Freshwater eutrophication (g P-eq)	0.0283	0.0715	0.0602	0.127
Marine eutrophication (g N-eq)	0.233	0.0842	1.83	0.116
Terrestrial ecotoxicity (kg 1,4 DB-eq)	0.00177	0.000230	0.00312	0.00699
Freshwater ecotoxicity (kg 1,4 DB-eq)	0.00151	0.00071	0.00359	0.00616
Agricultural land occupation (m^2.a)	0.239	0.402	0.445	0.458
Fossil depletion (kg oil-eq)	0.0195	0.0317	0.0334	0.0667

Mango and small citrus had both lower marine eutrophication (around 0.1 g N-eq), compared to apple (above 0.2) and peach (1.8). This was explained by the use of IPCC nitrate emission factors: being nil under drip-irrigated crops in semi-arid climate as mango from Brazil and small citrus from Morocco but reaching 30% of nitrogen inputs for crops under temperate climate as apple and peach from France.

Regarding terrestrial and freshwater ecotoxicity, mango showed for both the least impact followed by apple and then by peach and finally by small citrus far above. The great ecotoxicty impact for small citrus was essentially due to the use of Chlorpyrifos-ethyl for controlling California red scale in small citrus orchards in Morocco. This molecule has a very high toxicity potential and does not have efficient alternative up-to-date. The low ecotoxicity impact for mango was probably also the most uncertain of all since it relied on a small sample of farms. Furthermore, although the use of highly toxic molecules, such as Cypermethrin, in mango orchards had been orally reported we could not find evidence of such treatment in our sample of farms. This would definitely warrant further confirmation through survey across a wider sample of farms.

3.2. Comparison with published references

We could not find complete LCA studies for peach and mango. For apple and small citrus, our results were in the same range as results from the literature for GWP, Energy use, eutrophication and acidification (Table 3). For toxicity impacts, we only had one reference for each product to compare with. Overall, results were of the same order. Overall, the literature references confirm the least impacts of apple compared to citrus at farm-gate. This can be explained by higher nitrogen inputs and energy use for irrigation in citrus production associated to lower yields compared to apple.

4. Discussion

4.1. Farm-gate environmental impacts for 4 fruits consumed in France calculated with the Agribalyse® methodology

Until recently, fruits consumed in France had never been assessed with the LCA methodology. LCA references were therefore crucially needed to feed the eco-labeling program and debate on food products. Two locally-grown and two imported fruits were evaluated with the LCA methodology following a consensual method as dozens of other French agricultural products. This constituted an important step forward. At farm-gate these studies confirmed the greater impacts of citrus compared to apple mainly due to intensive practices associated to lower yield. It also produced novel references for peach and mango for which no LCA studies could be found worldwide. Contrary to most LCA studies on perennial products, in the Agribalyse® methodology the whole perennial crop cycle was modeled which represented an important and systematic progress. Beyond these important achievements, one should remind that in the Agribalyse® program the system boundaries were set at farm-gate which represents a limitation to properly compare imported with local fruits. The first reason relates to the exclusion of the transport of imported fruits to their final market which can represent important impacts. The second limitation is the non-inclusion at that stage of the quality requirements on fruits for export. From the total yield at farm-gate, only a fraction will have the required quality and should be allocated most of the impacts due to its higher economic value compared to the local quality fruits. Locally-grown (French here) fruits also show different qualities but will all end up on the local market. Moreover, in Agribalyse® the functional unit used was the kg of raw fruit while another important aspect for comparing fruits is the actual number of servings per kilogram of fruits also corresponding to their edible part (Ingwersen 2012; Cerutti et al. 2013). Thus, the rules for a proper comparison of imported versus locally-grown fruits need to be analyzed and formalized. At least the system

Table 3. Global warming Potential, non-renewable energy, eutrophication, acidification and toxicity impacts from different LCA studies for fruits and for this study. Results are expressed per kg of raw fruit at farm-gate.

Reference	Selected Product	Country	GWP (kg CO_2-eq)	Non-renewable energy (MJ)	Eutrophication (g PO_4-eq)	Acidification (g SO_2-eq)	Human toxicity (kg 1,4-DB-eq)	Aquatic freshwater ecotoxicity (kg 1,4-DB-eq)	Terrestrial ecotoxicity (kg 1,4-DB-eq)
Mila i Canals et al. (2006)	Integrated apple	New Zealand	0.04 – 0.095	0.41 - 0.7	n.a.	0.3 – 0.8b	-	-	-
Mouron et al. (2006)	Integrated apple	Switzerland	0.083	1.2	0.134	0.809b	-	-	-
Alaphilippe et al. (2013)	Conventional apple	France	0.032-0.038	0.44 - 0.51	0.23 – 0.33	0.20 – 0.23	0.012 – 0.014	0.005 – 0.010	0.002
Sanjuan et al. (2005)	Integrated orange	Spain	0.22 – 0.28	-	1.95	0.07 – 0.09	0.620	-	0.0043 – 0.0054
Beccali et al. (2009)	Lemon	Italy	0.155	2.33	0.636	0.994	-	-	-
	Orange	Italy	0.217	3.42	0.905	1.387	-	-	-
Trydeman Knudsen et al. (2011)	Conventional orange	Brazil	0.112	1.265	0.99	1.1b	-	-	-
Lo Giudice et al. (2013)	Integrated blood orange	Sicily	0.089	1.932	-	-	-	-	-
Pergola et al. (2013)	Conventional lemon	Sicily	0.12	2.85a	-	-	-	-	-
	Conventional orange	Sicily	0.13	2.87a	-	-	-	-	-
Ingwersen (2012)	Pineapple	Costa Rica	0.155	1.45	-	-	-	-	-
This study	Conventional apple	France	0.068	1.1	0.267	0.547	0.053	0.053	0.0038
	Conventional peach	France	0.168	2.5	1.27	1.83	0.119	0.135	0.0115
	Conventional small citrus	Morocco	0.269	3.3	0.679	2.08	0.107	0.733	0.0176
	Conventional mango	Brazil	0.139	1.5	0.49	1.64	0.079	0.061	0.0068

a: based on Namdari et al (2011), these figures include both renewable and non-renewable energy sources but Pergola et al (2013) explained that the renewable energy share is reduced to 5% in conventional systems. b: presented figures correspond to EDIP97 for Mila i Canals et al. (2006) and Trydeman Knudsen et al (2011) and to Heijungs et al. (1992) for Mouron et al. (2006) being probably overestimated compared to CML 2001 acidification results.

boundaries should be extended beyond the farm-gate to include transportation phases, fruit quality (including edible part) and allocation issues between the different fruit qualities. Regarding the method used for estimating field emissions, the most consensual and up-to-date ones were chosen which represented an important progress. However, the very generic emission factors used (such as IPCC or EMEP-CORINAIR ones) are not particularly valid for perennial crops under semi-arid climate. This represents definitely an important margin of progress and perspective for research. Finally, although water deprivation can represent a key environmental problem for fruit production, this indicator could not be included in the Agribalyse® program and this should be completed in upcoming studies.

4.2. Design of representative systems

One key difficulty of the Agribalyse® objectives was the design of representative systems in terms of technology, time and space. Important discrepancies between situations were faced in terms of data quality and availability. Where statistical average systems could be calculated for some products, others such as fruits were mostly evaluated through expert-based scenarios (apple, peach, small-citrus) or small samples of farms (mango). If expertise can be satisfactory for estimating most inputs and agronomic data, it is insufficient to capture the actual shares of pesticides used across a population. Knowing that certain active molecules have very high toxicity potentials, not having this statistical representation constitutes an important bias in the assessment of an average system. Using a small farm sample for mango proved even weaker in its capacity to capture the diversity of pesticide treatment practices and results for toxicity impact categories for mango should be seen as very uncertain. For French fruits, an additional effect has to be reported. Due to the French ECOPHYTO program aiming at reducing drastically pesticide use in France, most toxic molecules have been banned since 2009 (last year of the period covered by the Agribalyse systems). This means that a lot of molecules of pesticides used in our systems for apple and peach are no longer certified and used. Finally, it is important to keep in mind that designing representative practices over a period of 25 years represents a contradiction in itself and a difficult challenge especially for pesticide treatments which follow constantly changing rules.

5. Recommendations and conclusions

The Agribalyse® program permitted the production of a vast and consensual LCI database for French agricultural products including 4 fruits. This was definitely an important step forward. Effort should be continued to improve the representativeness of the average systems, develop the LCA beyond the farm-gate, include water deprivation and improve the methods for estimating field emissions under perennial cropping conditions in the South.

6. Acknowledgements

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Integrating social and economic criteria in the carbon footprint analysis in sheep dairy farms

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ABSTRACT

Climate change mitigation has recently become an important goal in our society. It is one the three objectives in the developing of last set of changes in the Common Agriculture Policy for the European Union. LCA methodology is able assess environmental impacts of food production along the food chain. Within LCA studies, carbon foot printing has becoming a popular indicator to communicate results to consumers, producers and all stakeholders involved in food sector. Sheep production is important activity in Southern Europe. Intensification of sheep farms is happening according to the global process of intensification of agriculture and livestock production in general and that involves, the decrease of people who are living in rural areas, the industrialization of the agriculture sector and this is a risk for local and small-scale producers. This paper presents results for emission intensity of two sheep milk systems in Northern Spain using different perspectives: productive (from 2.11 to 5.35 kg CO_2eq/liter of ECM), economic (from 0.84 to 13.4 kg CO_2eq/Net Margin), human resources (from 39,142.34 to 314,799.5 kg CO_2eq/Manpower Unit), and land occupation (from 964.11 to 6,882.43kg CO_2eq/ha). Lower values of emissions have been found for intensive farms per liter of FPCM and highest ones per land occupation, or manpower unit. Although improving efficiency and lowering costs are key factors reduce environmental impacts of food production, a holistic point of view is imperative to consider; and economic and social criteria need to be included in LCA assessments.

Keywords: Carbon footprint, Sheep farming, Multicriteria assessment.

1. Introduction

Climate change is one of the main issues that the livestock sector has been coping with recently (Gill et al, 2010), largely due to the importance of Greenhouse Gas Emissions (GHGs) coming from primary sector, especially from enteric fermentation of ruminants. Overall, 14.5 % of all human-induced emissions are produced by livestock sector (Gerber et al., 2013). Small ruminant's environmental impact is 10% of total methane emissions, and 18.83% of N_2O from manure management. (Gill et al, 2010).

LCA methodology has emerged as a tool useful to monitoring processes and identifies hotspots in different environmental categories, most of them focusing in food production (Flysjö et al, 2011). Global warming is one of them. Carbon footprint (CF) is becoming a popular indicator, considering its facility to understand, and communicate results from consumers and society in general.CF express intensity of kg of CO_2 per unit of product. Nevertheless, CF can generate conflict in other categories inside environmental quality indicators depending how results are reported, where high yield farms could have less emissions per unit produced. For this reason some knowledge is needed regarding the use of CF as an indicator for wider environmental impacts from food products (Röös et al, 2013).There are studies in biography which question if it's a good option increase milk yield per animal to decrease GHGs emission (Zehetmeier et al, 2012).

There is an urgent need to consider social and economic aspects in LCA studies (Flysjö et al, 2012). In addition to global warming impacts, food production also affects the environment considerably in many other ways (Röös et at, 2012) like land and resources demand (Garnett, 2014) .For the other hand, the concept of multifuncionality has become stronger (van der Ploeg & Roep, 2003; Van Huylenbroeck et al, 2007) to attribute more functions to agriculture, a part of providing food and fibres to society. The non-market functions of primary sector have been one of the midpoints of the future of the Common Agricultural Policy (European Commission, 2010) and it one of the justification with the link between the subsides with greener production systems (Matthews, 2013). Nevertheless, socio economic context which LCA results need to be situated (Edwards-Jones, et al, 2009) and different metrics to account GHGs could significantly alter the results and ranking (Reisinger & Ledgard,2013)

The overall aim of this study is to describe GHGs emissions with different functional units in order to extend the system boundary including social and economic factors to the classic environmental assessment. Carbon footprint is a useful conduct for analyzing the potential contribution of a product to climate change, but only efficiency approach (milk production) is taken into account.

This work has the following start point questions,
- How does the functional unit influence in the result of CF?
- Can we use metrics to influence in the interpretation of the results?

2. Methods

2.1. Case study

Sheep production has relevant importance in South Europe (de Rancourt et al, 2006) and in the Basque Country becomes a key sector in production of cheese (Ruíz et al, 2011) in semiextensive systems traditionally with an importance of pasture uplands during summer season. To tackle with the objective of this study, 12 sheep milk farms in the Province of Álava (Basque Country, Northern Spain) have been analyzed. All data was primary taken for SERGAL (advisory farmers association) for year 2011.The study involves this work required, more deeply data than other classical LCA studies, involving environmental, economic and social information of each farm through personal surveys and several visits every month.

Table 1. Main technical characteristics of farms analyzed in this study.

	Minimum value	**Maximum value**
Land occupation (ha)	29.1	229.1
Average annual temperature (ºC)	10.2	12.1
Workers (number)	1	5
Milk ewes (average population)	108	835.1
Annual production (liters of milk)	20,946	233,267
Liters/ewe year	109.1	399.4
Lambs (annual sales)	134	424
Lamb sold/ewe	0.4	1.16
% time grazing	0%	52.7%
Concentrate (kg/year)	29,185	388,495
Fodder (kg/year)	5,880	237,560
Oil(liters/year)	1,269.2	18,478
Electricity (kwH/year)	2,350	37,431.1
Mineral fertilizer (kg/year)	0	61,712.4
Prize milk (euros/liters milked milk)	0.77	2.4
Prize lambs (euros/kg lamb)	3.06	5.08

The main differences visible in farms are the high production of intensive farms (average yield per sheep 325 liters/year) with more traditional farms (average yield per sheep 110 liters/year); housing of intensive management flock (0% grazing per year) comparing with traditional flocks (approx. 48% grazing in natural grasslands); and price of sale of milk from 0.77 euros/liter at farm gate to those farmers who sell milk to industry to 2.4 euros/liter of milk to those who make cheese in their own farms and sell directly to consumers in local markers.

CF is been calculated following guidelines of PAS2050 (BSI, 2008) and IPCC criteria for livestock and soil emissions (IPCC, 2006).

2.2. System boundary

In order to integrate social and economic dimensions into LCA studies, environmental boundary has been integrated with social and economic boundaries. The system used in this study (Figure 1) coverts from cradle to farm gate. It includes emissions arising from manufacture and distribution of farm inputs ,the use of energy on the farm (fuels and electricity), the GHG emissions from livestock and their excreta, and emissions from soils related to fertilizer use and manure management. Nevertheless, environmental boundary does not take into account economic and social aspects as prices of inputs and sale prices, land occupation of the flock, as well as manpower units and relation of the farms with the environment. Thereby not only environmental resources are included in the assessment.

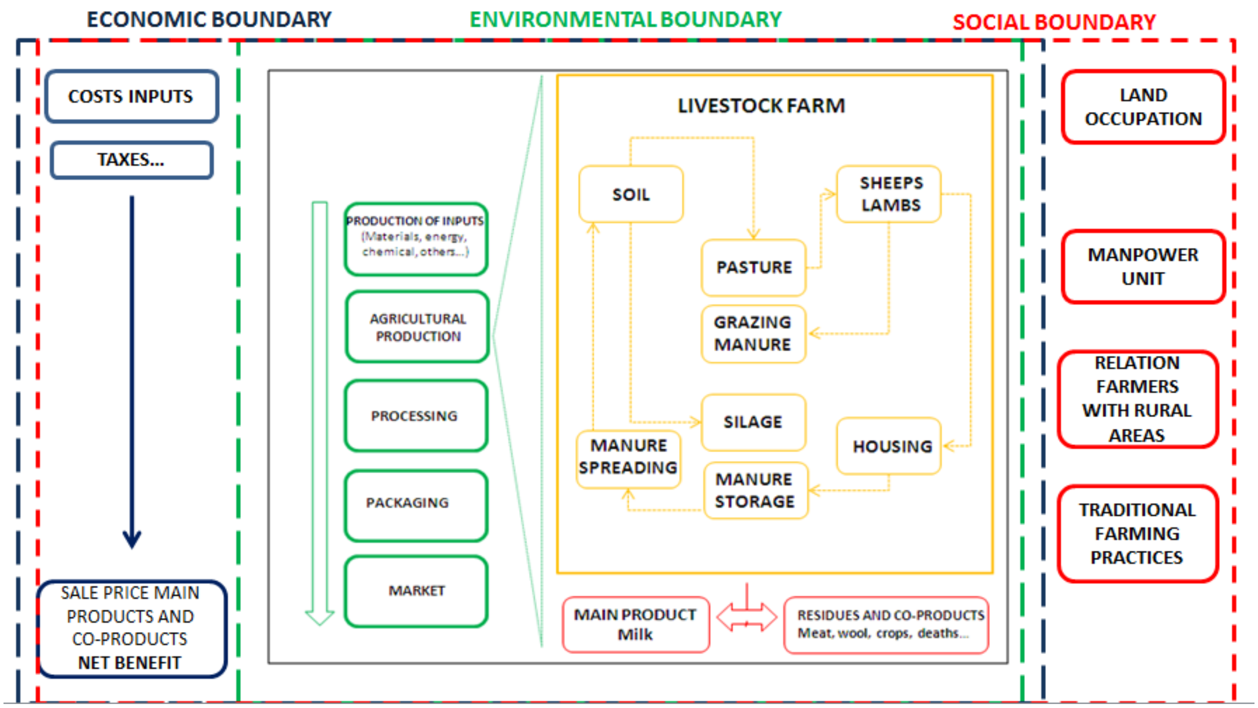

Figure 1. System boundary.

Emissions on farms and off farms (from production and manufactured of inputs) have been included in the breakdown GHG fluxes considered. According with system boundary, Table 2 shows these sources of emissions as well as the gases involved in each study and the source of the emission factor used. All the emissions are given in kg CO_2 equivalent, in accordance to the global warming potentials (GWP100) available from IPCC (2007).

Table 2. Source of emissions and models used in the study.

Sources of emissions	Gas	Source	OFF FARM/ON FARM Emissions
Enteric fermentation	CH_4	Torres et al (2004)	ON FARM
Manure management	CH_4	(IPCC, 2006)/MAGRAMA (2011)	ON FARM
Manure management	N_2O	IPCC 2006	ON FARM
- **Direct N_2O emissions**		Equation 10.25	
- **Indirect N_2O emissions**			
- **Volatilization**		Equation 10.26, 10.27	
- **Leaching**		Equation 10.28, 10.29	
Emissions from managed soils	N_2O	IPCC 2006	ON FARM
- **Direct N_2O emissions**		IPCC 2006, Equation 11.1, 11.2, 11.3, 11.4, 11.5, 11.6, 11.7, 11.8	
		IPCC 2006, Equation 11.9, 11.10, 11.11	
- **Indirect N_2O emissions**			
Emissions from liming	CO_2	IPCC 2006 Equation 11.11	ON FARM
Emissions from urea fertilization	CO_2	IPCC 2006 Equation 11.12	ON FARM
Energy use of farm (fuel and electricity)	CO_2	GES´TIM & IBERDROLA	OFF FARM/ON FARM (combustion)
INPUTS (feed, fertilizers, pesticides...)	CO_2	GES´TIM	OFF FARM

2.3. Functional unit and allocation

The functional unit (FU) is the unit which all GHGs emissions are reported relative to it. The FU is a measure of the function of the studied system and it provides a reference to which the inputs and outputs can be related, for that reason is a key element in the later work of reading the results. Traditionally, LCA studies on milk production use "1 kg of milk at farm gate" as functional unit. Table 3 shows the four functional units proposed in this study to cover economic, social and environmental boundary for the global warming potential of sheep milk production.

Table 3. Functional units proposed in the study.

Functional unit	Unit	Aspect to study
Energy Correct Milk	Kg ECM	Productive. Efficiency
Hectares	ha	Land Occupation
Manpower Unit	Number worker	Human resources
Net Margin	Euros	Economic/profitably

- Energy Corrected Milk (ECM). Milk yield from dairy sheep was corrected according to Bocquier et al, (1993) at 1200 kcal/liter of sheep milk.

$$ECM = 0,071 * fat (\%) + 0,043 * protein (\%) + 0,2224 \quad \text{Eq. 1}$$

- Hectares. Number of hectares that has been used by the sheep flock for feeding purposed during a year. This functional unit is proposed to focus the importance of land occupation by grazing for livestock, especially in less favorable areas, as uplands.

- Manpower Unit. Number of employee people on the farms. This functional unit copes to human resources necessary for milk production.

- Net Margin (NM). Difference between the sold price of outputs and the cost of all the inputs necessary for production including taxes. Profit of the farmer.

As shown in Figure 1, sheep production has edible and no edible products as outputs. Economic allocation was used to allocate GHGs emissions between milk, lamb and wool production. The reason of using economic allocation in the study is that in this case study, the main purpose of these farms, it is the milk sheep production, and lambs and wool are only co-products relative to the main production.

3. Results

Total emissions per ha, per Net Margin, per kg ECM produced and per Manpower unit are presented in Table 4 in maximum, average and minimum values.

Table 4. CF according with functional units (Minimum, average and maximum values).

Carbon footprint	Minimun values	Average values	Maximun values
Kg CO2-Equivalente / ha	964.11	3,190.75	6,882.43
kg CO2-Equivalente / NM	0.84	5.13	13.40
Kg CO2-Equivalente / Manpower Unit	39,142.34	131,309.69	314,799.50
kg CO2-eq/kg ECM	2.11	3.35	5.35

Using kg ECM corrected milk as FU, farms with high production (323 liters/ sheep year), will have lower CF. Normally these farms, are farms which lower land occupations, and with higher kg CO_2 eq/ha.

Figures 1, 2, 3 and 4 presents the correlation between emissions per functional unit and that functional unit, to highlight that depends in the functional unit chosen the way of read the results can be complete different.

Correlations between CF and functional units have been studied. There are no high correlations in any of the functional unit chosen in this study. Moderate negative correlation R^2 =-0.65 per ha (surface) and R^2 = -0.62 when Net Margin is the functional unit. On the other hand, positive correlation R^2 =0.41 for manpower unit, and high-moderate positive correlation, R^2=0.89 when liter per ewe is functional unit. Bigger differences can be seen between kg CO_2 per kg ECM and kg CO_2 per ha. If results are only given per kg ECM produced intensive production have lower emissions per functional unit, but these farms have higher emissions per land occupation.

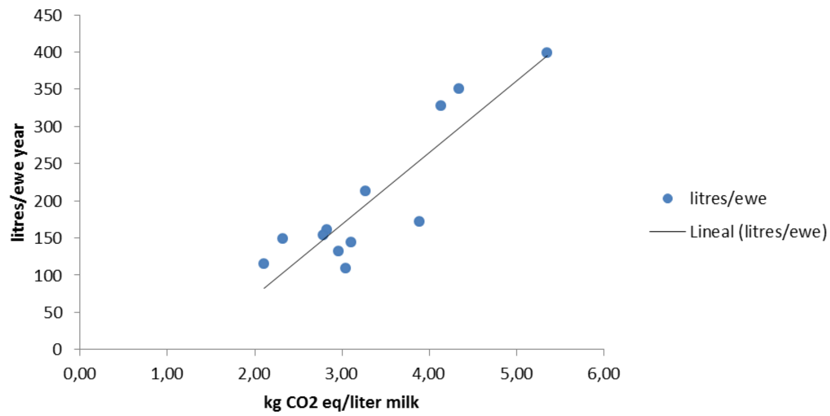

Figure 1. Relation between CF and milk annual production per sheep**.**

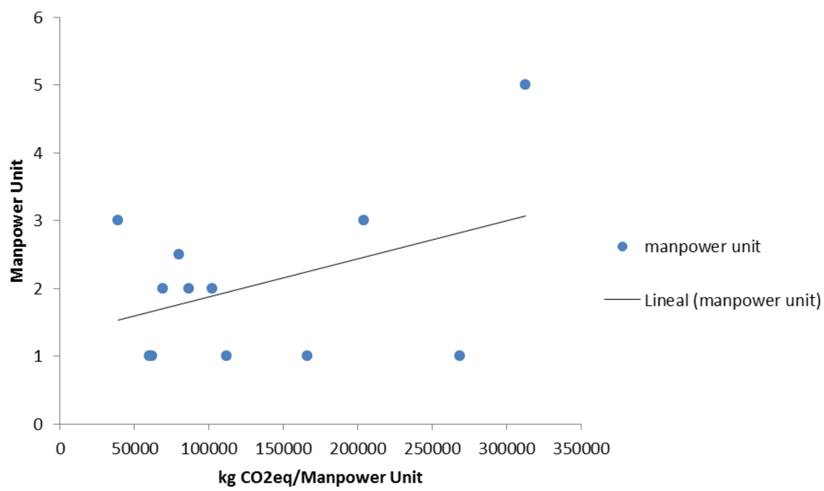

Figure 2. Relation between CF and Manpower unit.

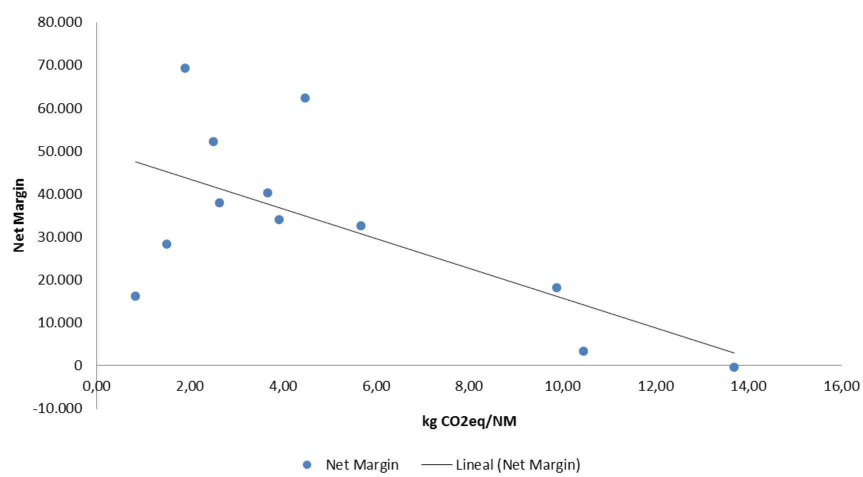

Figure 3. Relation between CF and profitability.

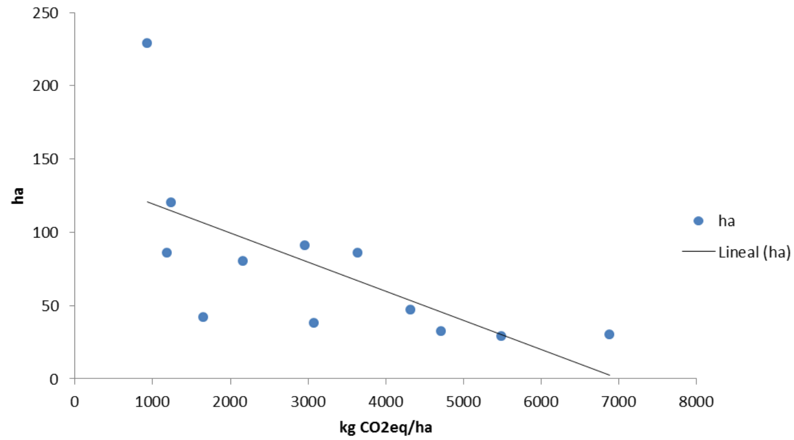

Figure 4. Relation between CF and land occupation.

4. Discussion

CF gives a number that means the contribution of one product to climate change. Table 4 shows results to the farms studied, emissions of 1 kg ECM are from 2.11 to 5.35 kg CO_2 eq/kg ECM. These results are similar to other studies, as Opio et al. (2013), Weiss and Leip (2012). If we only take into account this indicator as criteria of environmental assessment, we would choose the farm with lower CF as the most climate change friendly. In this example, the farm with lower CF corresponds to a very intensive management farm; totally different to the traditional sheep milk production in The Basque Country. Meanwhile, the one with higher CF corresponded with a traditional farm with grazing as important feed resource for the flock, typical from uplands in this territory. It is necessary not only think in productive functional unit (not only efficiency could be the criteria) and CF message could be misunderstood. Table 5 displays how farms can have the maximum and minimum value depends in which functional unit we take into account. For example, farm number 11 has one of the maximum values of CF per kilogram of ECM, but in the other hand, has one of the lowest values of emissions per hectare. This is an example of how traditional farms, less productive than the intensive could have higher intensity of emissions if we only focus in productive functional units.

Table 5. Benchmarking of the 12 farm studied depending in the different functional units.

	kg CO_2-eq/kg ECM	kg CO_2-Eq/ NM	Kg CO_2-Eq / Ha	kg CO_2-eq/Manpower Unit
Farm with maximum value	Farm 12	Farm 7	Farm 1	Farm 1
	Farm 11	Farm 9	Farm 10	Farm 9
	Farm 6	Farm 1	Farm 4	Farm 11
	Farm 7	Farm 2	Farm 12	Farm 7
	Farm 5	Farm 3	Farm 3	Farm 12
	Farm 4	Farm 6	Farm 11	Farm 8
	Farm 8	Farm 8	Farm 2	Farm 6
	Farm 10	Farm 12	Farm 6	Farm 4
	Farm 9	Farm 10	Farm 9	Farm 10
	Farm 1	Farm 4	Farm 5	Farm 3
	Farm 2	Farm 5	Farm 7	Farm 2
Farm with minimum value	Farm 3	Farm 11	Farm 8	Farm 5

Fig 5 presents kg CO_2/kg ECM and kg CO_2/ha for the farms studied. Dots represent kg CO_2/kg ECM and bars represent kg CO_2/ha. In general the one with are higher with one criteria are the lowest values for the other.

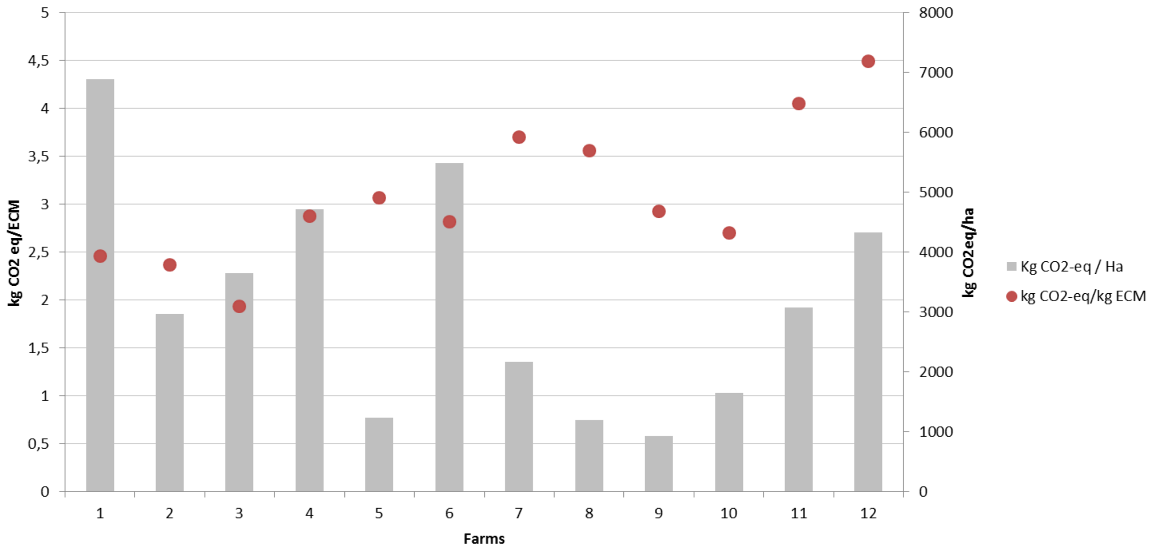

Fig 5. kg CO_2/kg ECM and kgCO_2/ha in the 12 farms studied.

5. Conclusion

Although improving efficiency, lowering cost are key factors on improve environmental impacts of food production, a holistic point of view is imperative to consider; and economic and social indicators are needed to include in LCA assessment.

CF is a useful indicator to monitor emissions by individual farms. Nevertheless, when results between farms are compared, these numbers could be misunderstood, particularly when high differences of efficiency occur within them. This work presents how using different number to express emissions, using different FU could arrive to different results, depending on the socio-economic context.

LCA studies need to take into account other aspects beyond yield to enlarge system boundary and strengthen role of alternative production systems (organic, traditional productions) and several functions given to primary sector nowadays.

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An innovative methodology combining Life Cycle Assessment of a product with the assessment of its Quality; case of the French vineyards

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ABSTRACT

In the wine sector, professionals want to reduce the environmental impact of their production but without taking the risk of reducing the quality of the grapes produced and the yield. The present study consists in combining environmental and quality assessment in order to take decisions about the vineyard management strategy. A multi-criteria assessment method will be used to combine those both aspects. But first, the LCA methodology must still be improved in viticulture, in particular for taking into account the carbon cycle (with carbon storage). As LCA results are potential impacts, the quality assessment needs to show also potential results instead of measured data as it is done today. A predictive model for quality evaluation, based on the vineyard management and the natural environment is, hence, being developed in this project.

Keywords: Life Cycle Assessment, multi-criteria analysis, quality, viticulture, practices, grape

1. Introduction

French government has developed a policy on sustainable development that includes a 50% reduction on the use of pesticides between 2008 and 2018. Viticulture sector is concerned about this policy because even if it represents only 3.7% of the French UAA (Utilized Agriculture Area) it uses 20% of pesticide (in kg) consumed in the country (Aubertot et al. 2005). The use of pesticide isn't the only concern of winegrowers. They also wish to reduce on a global way, the impact of their practices on the environment.

The vineyard technical management route (TMR)[1] implemented by the grower, soil and climate factors are the main determinants of quality and environmental impacts of the grapes. These factors have complex and sometimes opposite effects on environmental impacts and quality of grapes.

First of all, LCA methodology for viticulture (Renaud-Gentié et al. 2012) needs to be completed in particular by taking into account the carbon cycle (with carbon storage). Results of LCA that are presented through impact categories will be selected, combined or weighted for inclusion in a multi-criteria analysis to be compared with the quality assessment.

Secondly, the methodology of grape quality assessment has also to be adjusted for insuring comparison with LCA and for joining them together. As LCA permits predictive scenario analysis through potential impacts calculation, the quality assessment needs to give also potential results for predictive analysis besides the measured data that are available today. A predictive model for quality assessment, based on the vineyard management in one hand and soil and climate in the other hand is, hence, being developed in this project.

Finally, results of LCA that are composed by midpoint impact categories and results from the quality evaluation of grapes (which also gives many different variables) will be assessed jointly through the construction of a decision tree.

Methodologies are applied for a white grape variety on five vineyards plots, which represent the diversity of practices of the Middle Loire Valley (France) dry wine vineyards plots in Protected Designation of Origin (PDO) (Renaud-Gentié et al. 2014).

2. Methodological issues

The present study considers all activities at the field from the plantation to the uprooting of vines and by taking into account one year of full production. The period considered for the production year is from the end of the last harvest until the end of the harvest of the year considered. Environmental impacts of phases of planting and

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[1] Technical Management Routes (TMRs): logical successions of technical options designed by the farmers (Renaud-Gentié et al. 2014)

uprooting of vines are amortized on the life of the vine. What happens after the harvested grapes have left the field is not considered in this study. As a consequence, winemaking process isn't considered, just as wine storage, bottle production and distribution are described in (Renaud et al. 2011).

The system boundaries include the equipment, energy, and water consumed during the viticulture period. The transportation of workers from the headquarters to the plot is considered and also the transport of all inputs.

The joint assessment method for combining Quality Assessment with LCA is being developed by Christian Bockstaller and is named CONTRA (Transparent Construction of Decision Trees) (Kaueffer 2013). It is an aggregation method based on knowledge and preferences that integrates fuzzy logic (Bockstaller et al. 2013).

LCA analysis is performed using SimaPro software (SimaPro 8). The inventories of practices come mainly from the analysis of the winemaker's practices. They are completed by data from the industry, climate data of the region studied, soils characterizations, the Ecoinvent database (version 3) and national database.

The Quality of grapes is evaluated from the incorporation of data about the practices, soil and climate of the year considered into a Quality assessment model that is being developed in this research program. This model is based on Partial Least Square (PLS) Regression 2 (Sang and Lee 2009) (Reinikainen and Höskuldsson 2007) applied on a data set of plots studied for four years in the region of the present researches. The resulting model will be validated after comparison with actual measurements of quality study plots. The approach used for developing the joint evaluation is explained through a scheme shown in Figure 1.

The experimental network concerns the Chenin Blanc grape variety, which is one of the major cultivars of the Middle Loire Valley for white wine production. Five plots representing the diversity of vineyard management strategies of the central Loire Valley PDO vineyards compose the network (Renaud-Gentié et al. 2014).

Two functional units are considered for this study; first the surface of 1 ha of a producing vineyard. This functional unit will enable us to compare the systems from the land occupation point of view. The second functional unit is the kg of grapes produced.

3. Discussion

A part of this study is to continue the adaptation of LCA on viticulture from the research lead off by Christel Renaud-Gentié (Renaud et al. 2012). Most of these adaptations consist in taking into account the carbon cycle in the assessment including carbon sequestration (Rabl et al. 2007; Helin et al. 2013).

First, the vine leaves falling on the soil each year are biodegraded by the soil fauna and provides a part of minerals and nutriments that will allow the plant to continue growing. The balance is not null but there is compensation between the material consumed by the plant for the production of leaves and the carbon released by the plant when the leaves fall.

Secondly wood is produced during one year of production by the development of the vine cane. The main part of this wood is cut every year after harvest. The wood can be mowed, left on the soil and directly biodegraded by the soil fauna, but it can also be burnt as energy source or be collected and processed by a waste treatment plant.

The quantity of wood produced yearly represents a small quantity compared to the quantity of wood available at the end of the vineyard life while the vine plants of a plot are grubbed up. At its end of life, this wood is mostly burnt through house warming or collected and processed by a waste treatment plant. In this case represents a much bigger quantity of carbon potentially released and suddenly a more significant environmental impact depending on the end of life destination of the wood. In addition, while plot are grubbed up, there is a release of carbon associated with the destruction of the cover crop, if present, at the time of pulling.

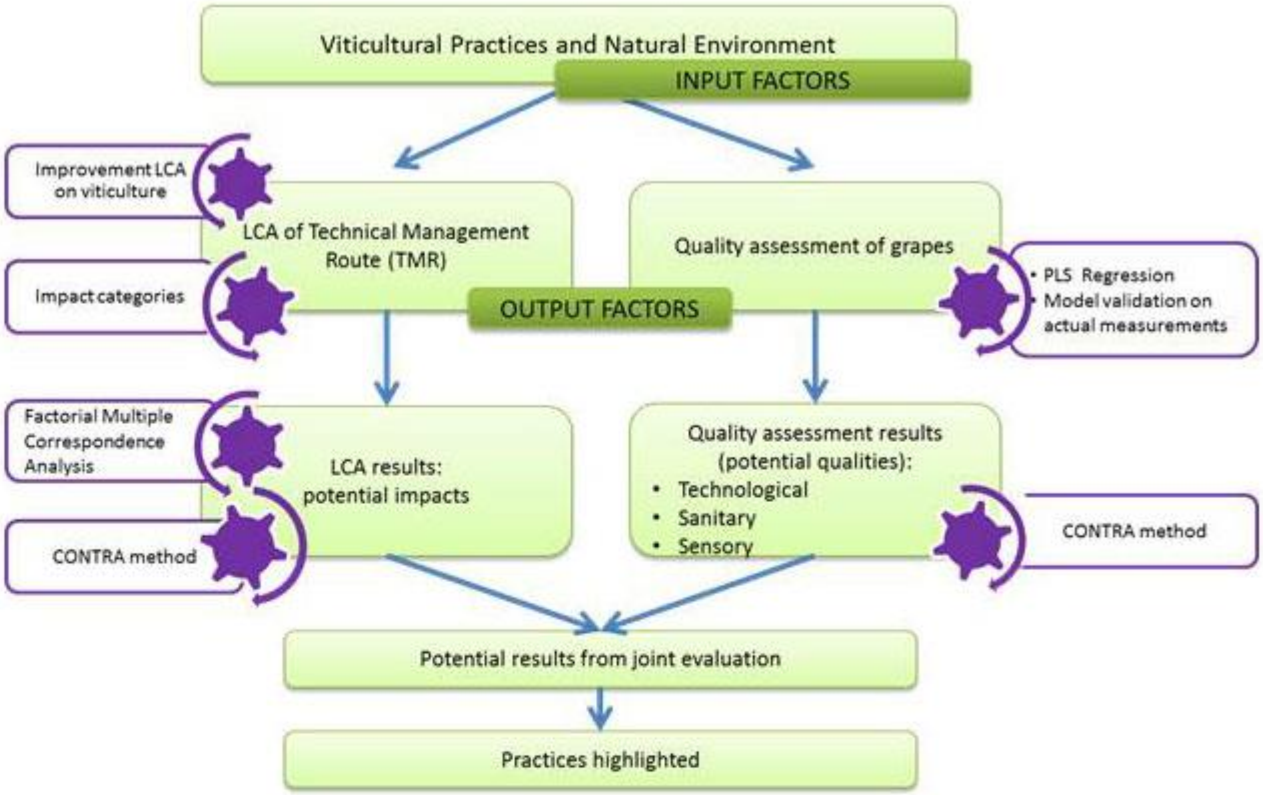

Figure 1. Overall scheme of the joint assessment process.

Concerning the environmental assessment, it should also be noted that some environmental indicators may not tie into the problems of the viticulture sector. That's why indicators used for assessment will be those, which represent an issue for the professional sector like Global Warming potential, acidification potential, toxicity indicators or even land use.

For ensuring the combination of LCA with Quality assessment a predictive model of the grape Quality based on practices and environmental factors (soil, climate…) is being developed. Most factors that influence the quality of grapes are taken into account. The notion of Quality can be defined in many different ways. In this study, the goal is to define a Quality assessment that would help the winegrower to transform its grapes into the best wine he could expect to have. To do so, Quality is assessed through the technological, sanitary and sensory evaluations of grapes. The sanitary assessment is the visual estimate of the decay rate of grapes. Technological qualities are evaluated by the sugar level, phenolic compounds by spectrophotometry (SO 280 and DO 420), pH and acidity of a representative grapes berries sample. Sensory assessment considers aromas, sugar, acidity, the grape berries texture and other properties of a representative grapes berries sample. The panel who evaluates the grapes berries is composed of industry professionals who taste the berries up to a maximum of 48 hours after collection.

For each of those components of the Quality, several parameters have been defined and are measured.

Most of those parameters have been registered in past years on different plots of Chenin Blanc in the Loire Valley, and kept in databases.

From those data previously mentioned, a mathematical relation is being built between the inputs (practices, climate and soil) and the outputs (sensory, sanitary and technological Qualities). It's important to notice that there also are interactions between input data and it's the same for output data.

Partial Least Square Regression 2 (Reinikainen and Höskuldsson 2007) is able to find relations between input and output data while there is interaction between factors. The results of the PLS Regression 2 (Reinikainen and Høskuldsson 2007) will be evaluated by a group of experts that will adjust its structure according to their expertise. They will define priorities between each input: practices, climate and soil and its degree of influence on each component of the quality (Table 1).

Indicators	Components	Type of assessment
Global Warming Potential		Results of Life Cycle Assessment
Ozone formation		
Acidification		
Eutrophication		
Human Toxicity	Human Toxicity	
Terrestrial Ecotoxicity	Ecotoxicity	
Aquatic Ecotoxicity		
Resources consumed	consumption of natural resources	
Total water use (blue water)		
Non renewable, fossil		
Yields grapes	Yields	Results of Quality Assessment
Yields musts		
Sugar	Technological analyzes	
Total acidity		
Malic acid		
Tartaric acid		
Assimilable nitrogen		
Optical density at 420 nm		
Optical density at 280 nm		
pH		
$d^{13}C$		
Decay rate	Sanitary analyze	
Hue bay	Sensory analyzes of berries and juice	
Firmness bay		
Amount of flesh on the pedicel		
Juiciness bay		
Aroma pulp		
Shredding of the film of bay		
Agressiveness of the film		
Color of the seeds		
Juice aroma		
Hue juice		
Sugar juice		
Juice acidity		

Table 1. Presentation of the decision tree for joint evaluation

Developing a predictive empirical model of quality may be not accurately reflecting the real phenomenon that occurs. The objective in this study isn't to reflect the exact reality but to get as close as possible from it for being able to model the consequences of a change in practice, taking into account environment factors. There are many interactions between the input factors of the model and those factors have numerous interactions that influence model outputs. Similarly, introducing the sensory evaluation of grapes quality is a challenging innovation as sensory analyzes must be adapted for each grape cultivar. Finally, there might be too many quality indicators to get the joint assessment; that's why they will have to be selected.

Thanks to the predictive method for the grape quality it should become possible to estimate the quality of grapes from the practices on the field, its soil and climate. It's important to notice that the model for evaluating the Quality of grapes from practices, climate and soil can be used only for the cultivar Chenin under the Loire Valley climate conditions. Indeed the Quality assessment is based only on this cultivar for the region considered.

At the end of the LCA and Quality assessments there are multi-results for each assessment. CONTRA can help joining both assessments but in this case there are too many results. For LCA results, a Factorial Multiple Correspondence Analysis (FMCA) will be applied on impact categories to keep only the main components of environmental impacts; avoid duplication of information and therefore reduce the number of impact categories.

The methodology CONTRA helps defining, with the user decision rules aggregation based on results from LCA and Quality assessment model.

4. Conclusion

The combination of LCA and grape quality assessments will allow understanding how to improve practices and select the ones, which prove to be the best trade-offs between quality and environmental performances.

It will be possible to analyze the input data that are responsible for the impact on the environment through one or more impact categories. And in the meantime we'll know their importance on grape quality. According to their importance on the combined assessments, advices will be given to the winegrower to point out the impact of his practices.

By working on five plots which are managed in completely different way, it will be interesting to identify practices that are able to improve both aspects of the assessment (environmental and quality) or that are able to decrease the environmental impact of the process without reducing the quality of the product. The positioning of each TMR evaluated will give a list of the vineyard management techniques responsible for results obtained. Thanks to that, advices would be given to the winegrower for adapting some of his practices for decreasing the environmental impact of his grape production without reducing or even with increasing the quality of his grapes.

The research on grapes led us to believe that those methods could also be applied on other agricultural productions; perennial products first but also annual products. It would also be interesting to even include this multi-criteria assessment in or combine it with Sustainable Life Cycle Assessment.

5. Acknowledgements

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Considering the variability of farming practices improves the LCA of biodiesel from oilseed rape

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ABSTRACT

Nitrogen fertilization practices have a significant effect on the LCA results of biodiesel chains, which warrants reliable inventory data. In this study focused on the Lorraine region (eastern France), we established a typology of oilseed rape fields based on fertilization practices, and used the agro-ecosystem model CERES-EGC in lieu of generic emission factors to simulate the productivity and externalities associated with oilseed farming. The results were subsequently used to generate an LCA of biodiesel from oilseed rape. We also tested the effect of improved practices on the LCA results. In Lorraine, oilseed rape crops appeared to be frequently over fertilized compared to best management practices. Switching to improved practices with optimal fertilization has a potential to reduce the GWP of 1 megajoule of biodiesel by around 6 gr CO_2eq, against a total life-cycle of 43.9 gr CO_2eq.

Keywords: biodiesel, oilseed rape, farming practices, emissions modeling, LCA

1. Introduction

The development of the use of biofuels in the transportation sector in Europe is the consequence of policies aiming at enhancing energy security and mitigating climate change through the reduction of anthropogenic greenhouse gas (GHG) emissions. After the recognition of the unexpected effects of the development of first generation biofuels on agricultural prices and land-use, a set of sustainability criteria outlined in a directive dedicated to renewable energy must henceforth be respected (Ben Aoun et al. 2013). From 2016, the substitution of fossil fuels by biofuels should allow the saving of at least 50% of GHG emissions (European Commission 2009).

The standardized method of life cycle assessment (LCA) is currently widely recommended for evaluating the environmental balance of products. When applied to bioenergy chains, several factors may influence its outputs. Thence, LCA studies concerning the same biofuel type can lead to significantly different results (IPCC 2011). In addition to methodological choices such as system boundaries, functional unit and co-products handling, these variations are essentially due to differences in the life cycle inventory stage. Recent studies (Davis et al. 2013, Smeets et al. 2009) showed that data on the feedstock production stage account for most of this variability. According to the same studies, this arises from the fact that the larger the geographical scale of the LCA study (eg. a region or a country), the coarser the data that depend on local factors (soil type, climate conditions and farming practices). The latter include crop yields and GHG emissions, in particular nitrous oxide (N_2O). This translates into a high uncertainty band for the biofuels LCA results.

As a consequence, improving the quality of such LCAs requires a better consideration of local factors when estimating agronomic and environmental data. The use of agro-ecosystem modeling is one of the solutions proposed to avoid the use of generic, fixed emissions factors (EFs) with a worldwide scope as suggested in the Tier 1 methodology of IPCC guidelines (2006). Such an approach would make it possible to obtain site-specific and reliable values when estimating N_2O emissions from soils (Dufossé et al. 2013, Cherubini 2010).

The objective of this study was to reduce the uncertainty surrounding biofuels LCAs through the use of an agro-ecosystem model (CERES-EGC) to estimate the agricultural and environmental variables needed for establishing the life-cycle inventory. It was applied to biodiesel from oilseed rape, the main biofuel produced in Europe, in the context of the Lorraine region (eastern France). Given that organic and inorganic nitrogen fertilization is responsible for 90% of the life-cycle GHG emissions related to oilseed rape production in France (Cerrutti et al. 2013), the effect of improved fertilization practices on LCA results was explored.

2. Methods

2.1. Goal and scope

This study focused on improving the quality of the inventory phase for the LCA of biodiesel from oilseed rape, using agro-ecosystem modelling and farm surveys. Biodiesel LCA results were compared to those of LCA of fossil-based gasoline. The effects of improved fertilization practices on the initial biodiesel LCA results were also examined.

For reasons of data availability, this study only focused on biodiesel made from oilseed rape harvested during the year 2012 in the Lorraine region in France. A study on the whole France is currently in progress.

The evaluated system includes five main stages: oilseed rape production, delivery to a biorefinery, conversion to biodiesel, storage in the plant and distribution, i.e. well-to-wheel boundaries. One mega-joule of biodiesel is the functional unit retained for this study. Rape meal, acid oils and glycerin are co-produced during the biodiesel conversion stage. To share environmental burdens between biodiesel and its co-products, the energetic allocation was applied, as recommended in the renewable energy directive (European Commission 2009).

2.2. Life cycle inventories

2.2.1. Crop management and biomass conversion data

Information on farming practices were derived from survey data conducted on behalf of the French technical center of oilseeds and hemp (CETIOM). A typology of fields producing oilseed rape in Lorraine was established according to farmers' nitrogen fertilization practices. In this typology, we assumed that the amount of inorganic nitrogen fertilizers applied on oilseed rape depends on three key factors: whether organic fertilizers were regularly applied (i.e. at least once every four years) or not, whether organic fertilizers were applied before sowing, and whether a decision support tool was used or not. This allowed us to identify 10 different fertilization practices (Table 1).

Table 1. Typology of oilseed rape management in Lorraine

Fertilization practice type	Regular fertilization	Organic fertilization before sowing	Use of decision support tool	Percentage of occurrence in the region
1	No	No	No	13.4%
2	No	No	Yes, well-respected	1.1%
3	No	No	Yes, non-respected	22.9%
4	No	Yes	No	6.3%
5	No	Yes	Yes, non-respected	3.2%
6	Yes	No	No	5%
7	Yes	No	Yes, well-respected	0.6%
8	Yes	No	Yes, non-respected	13.4%
9	Yes	Yes	No	15%
10	Yes	Yes	Yes, non-respected	19.1%

The decision-support tool was developed by CETIOM and calculates the optimal amount of inorganic fertilizer N to be applied in spring, based on a balance-sheet method. Therefore its use allows to avoid situations of both under and over-fertilization, which can largely influence crop yields and emissions of reactive N (Nr).

Particular attention was given to the form of organic fertilizer applied, as it can have considerable effects on Nr emissions (Dambreville et al. 2008). Still, for the sake of simplification only the most commonly-used forms were retained. Therefore, we considered that cattle manure was the only form of organic fertilizer applied to oilseed rape in Lorraine.

Data on machinery and inputs production (i.e. fertilizers, pesticides, fuel, etc.) were taken from the French data base AGRIBALYSE, recently developed by the French Environment Agency (Koch and Salou 2013). Data on oilseed rape transport and its conversion to biodiesel were taken from the Ecoinvent data base (v2.0). Those concerning biodiesel distribution were taken from a recent study carried out by ADEME (Biois 2010). Table 2 summarizes the data sources used for the life cycle inventory of biodiesel from oilseed rape in Lorraine.

2.2.2. Crop simulations

The agro-ecosystem model CERES-EGC can simulate yields, soil carbon dynamics and reactive nitrogen emissions including N_2O, as detailed by Gabrielle et al. (2006). To make possible the use of the model on larger scales while taking into account the variability of local conditions, a GIS database containing geo-referenced informations on the administrative borders, land cover, soil properties and climatic conditions was built. The corresponding layers of spatial information were overlaid to delineate elementary spatial units. These units represent a unique combination of soil type and climatic conditions to be used in the CERES-EGC simulations (Dufossé et al. 2013, Gabrielle et al. 2014).

Table 2. Sources of data for life-cycle inventory of the pathway investigated.

Stage	**Source**
Feedstock production	
Crop management	CETIOM
Crop yield	CERES-EGC simulations
Machinery & inputs production	AGRIBALYSE
Direct N2O emissions	CERES-EGC simulations
Indirect N2O emissions	CERES-EGC & IPCC 2006 guidelines
Other Nr losses	CERES-EGC simulations
Soil c dynamics	CERES-EGC simulations
Transport	Ecoinvent database
Conversion to biodiesel	Ecoinvent database
Distribution	ADEME

For each practice, daily simulated fluxes of direct Nr emissions were accumulated from the harvest of the previous crop to the harvest of oilseed rape. Indirect emissions of N_2O due to nitrogen leaching were calculated following the 2006 IPCC guidelines as 0.75% of the nitrate losses and 1% of ammonia and nitric oxide emissions, as simulated by CERES-EGC.

To simplify the regional modeling, we assumed an uniform distribution of management practices (as detailed in the above typology) over the whole region, i.e. assuming the occurrence of fertilization practices in each CERES-EGC simulation unit was the same as that on a regional scale. Therefore yields and externalities related to each fertilization practice have been simulated for each simulation unit. Then, results obtained from these simulations make it possible to estimate a representative regional yield and nitrogen emissions by weighting the simulations results of each practice by its percentage in the region (via Equations 1 & 2).

$$\sum_{i=1}^{10} y_i * perc_i = Y_{Lorraine} \qquad \text{Eq.1}$$

$$\sum_{i=1}^{10} e_i * perc_i = E_{Lorraine} \qquad \text{Eq.2}$$

With :
i : fertilization practice n° i
y_i : yield of the fertilization practice n° i
e_i : emissions related to oilseed rape production via the fertilization practice n° i
$perc_i$: percentage of the fertilization practice n° i in the region
$Y_{Lorraine}$: regional oilseed rape yield
$E_{Lorraine}$: regional emissions due to oilseed rape farming

2.2.3. Impact characterization and interpretation

LCA was conducted for each fertilization practice using Sima-Pro software package (v7.2). The environmental impacts were characterized with the CML 2000 method at mid-point level. The following impacts categories were analyzed: global warming, eutrophication, acidification, depletion of abiotic resources, photochemical ozone formation. Results were subsequently aggregated on the regional scale

3. Results

3.1. Current fertilization practices may still be improved

We used the CETIOM survey data to determine the quantities of inorganic nitrogen currently applied on oilseed rape for each fertilization practice described above. We found that oilseed rape producers apply about 158 kg of inorganic nitrogen per hectare in Lorraine on average.

We also calculated the recommended amount of inorganic nitrogen to be applied for each fertilization practice using the nitrogen balance method. Results show that inorganic nitrogen rates currently applied exceed the recommended ones (Table 3). Indeed Lorraine farmers tend to over fertilize oilseed rape: only less than 2% of oilseed rape fields receive the recommended dose of inorganic nitrogen.

3.2. Field emissions of reactive N

Table 4 lists the regional yields and emissions of reactive nitrogen for the various oilseed rape fertilization practices obtained with the ecosystem model CERES-EGC.

Table 3. Differences between current and recommended fertilization practices (in kg N/ha)

Fertilization practice type	**Percentage in the region**	**Current inorganic Nr fertilization**(1)	**Recommended inorganic Nr fertilization**(2)	**Fertilization gap** (1-2)
1	13.40%	160.35	117.03	43.32
2	1.10%	170.00	166.05	3.95
3	22.90%	156.15	122.75	33.40
4	6.30%	158.88	100.30	58.58
5	3.20%	176.75	96.29	80.46
6	5.00%	175.89	94.37	81.52
7	0.60%	165.00	159.50	5.50
8	13.40%	148.83	86.43	62.40
9	15.00%	156.18	82.89	73.29
10	19.10%	158.14	91.11	67.03
Average	100.00%	158.17	102.12	56.05

Compared to yields reported in the CETIOM survey, predicted oilseed rape yields in 2012 for the various fertilization practices were over-estimated by around 15%. As the model does not take accidents such us crop diseases or weeds into account, and since the year 2012 was marked by a significant occurrence of this type of accident in Lorraine region, these simulated yields can be accepted and the risk of a systematic modeling bias may be considered low. Also, harvest losses are not included in the modelled yields and may amount to the 15% discrepancy observed.

According to CERES-EGC, the highest direct emissions of N_2O occurred with fertilization practices in which organic nitrogen is applied before sowing of oilseed rape in the fall (i.e. fertilization practices no 4,5,9 and 10). For the practices involving only inorganic fertilization, direct N_2O emissions were much lower. Here, one should mention that for the various fertilization practices, direct N_2O emissions simulated by CERES-EGC are 2 to 3 fold lower than would be estimated when using the Tier 1 IPCC guidelines. In contrast, calculated indirect N_2O emissions were higher than what could be found using the IPCC methodology. This is due to the high nitrate losses simulated by the model.

Regarding optimal fertilization practices, results show that both direct and indirect N_2O emissions and nitrate decreased by 10 to 15% when recommended practices were simulated in lieu of current ones, whereas crop yields were less sensitive to these changes (Table 5)

3.3. Life-cycle impacts of biodiesel from oilseed rape chain

Table 6 reports the environmental performance of biodiesel from oilseed rape for the various fertilization practices adopted by farmers during the feedstock production phase in Lorraine.

Our results show that all impact categories selected in this study are sensitive to nitrogen fertilization practices, except for the ozone formation and abiotic depletion categories which remained unchanged regardless of the fertilization practice. Thus, Nr emissions, in relation to N fertilization have a significant effect on biofuels LCA results, whether from direct field emissions or upstream inputs manufacturing. This is in line with findings by Gabrielle et al. (2014).

Table 4. Nitrogen losses and oilseed rape yields predicted by CERES-EGC with current practices

Fertilization practice type	**Yield (Kg ha^{-1})**	**Direct N_2O (Kg N-N_2O ha^{-1}**	**Nitrate (Kg N-NO_3 ha^{-1})**	**NH_3 (Kg N-NH_3 ha^{-1})**	**NO (Kg N-NO ha^{-1})**	**Indirect N_2O (Kg N-N_2O ha^{-1})**
1	3077	0.59	26.03	22.14	1.31	0.88
2	3037	0.66	26.44	20.57	1.30	0.88
3	3074	0.58	26.04	22.16	1.31	0.87
4	3410	1.36	64.15	53.45	2.39	2.16
5	3453	2.01	106.99	101.86	3.30	3.87
6	3041	0.68	26.61	14.67	1.13	0.82
7	3103	0.62	26.29	14.86	1.15	0.81
8	3051	0.58	26.00	18.69	1.29	0.85
9	3446	1.83	96.77	94.61	3.04	3.39
10	3446	1.82	95.68	95.01	3.04	3.37
Average	3229	1.09	54.95	50.04	2.01	1.90

Concerning the global warming potential (GWP), all fertilization practices currently adopted by Lorraine farmers allow the whole biodiesel chain to emit less GHG than fossil diesel if land use change effects are not considered. On a regional scale, the GHG intensity of 1 mega joule of biodiesel from oilseed rape in Lorraine is about 43.9 gr CO_2 eq. The European sustainability criteria are thus not met because the 50% abatement threshold for GHG emissions is not reached.

Table 5. Nitrogen losses and oilseed rape yields predicted by CERES-EGC with optimal practices

Fertilization practice type	**Yield (Kg ha^{-1})**	**Direct N_2O (Kg N-N_2O ha^{-1}**	**Nitrate (Kg N-NO_3 ha^{-1})**	**NH_3 (Kg N-NH_3 ha^{-1})**	**NO (Kg N-NO ha^{-1})**	**Indirect N_2O (Kg N-N2O ha^{-1})**
1	2995	0.51	25.13	10.37	1.12	0.74
2	2954	0.55	25.11	23.06	1.20	0.87
3	2841	0.51	25.11	11.64	1.13	0.75
4	3344	1.25	59.09	37.63	2.18	1.87
5	3405	1.87	101.86	117.05	3.06	3.53
6	2850	0.50	25.33	3.90	0.97	0.68
7	3064	0.58	25.34	13.44	1.03	0.77
8	2857	0.48	25.33	4.50	1.06	0.68
9	3380	1.72	91.86	74,31	2.82	3.06
10	3388	1.73	91.79	75.63	3.83	3.07
Average	3090	1.01	52.48	36.18	1.81	1.63

Adopting best fertilization practices for oilseed rape production may significantly improve the environmental balance of biodiesel in Lorraine. Overall, the life-cycle GHG emissions of biodiesel may decrease from 43.9 to 37.6 g CO_2 eq per MJ of biofuel (Table 7). This translates as an abatement of 56% when substituting gasoline with oilseed rape biodiesel, and thus ensures compliance with European sustainability criteria.

Figure 1 compares the environmental performance of biodiesel from oilseed rape produced with current fertilization practices to biodiesel processed by rape produced with optimal fertilization practices. It shows that balancing the inorganic nitrogen fertilization to crop nitrogen needs could allow reducing GWP and eutrophication by 15%, while mitigating acidification by 35%.

Table 6. Environmental performances of 1 MJ of biodiesel from oilseed rape with current practices in Lorraine

Fertilization practice type	GWP (100 years) (g CO_2 eq)	Acidification (kg SO_2 eq)	Eutrophication (kg PO_4 eq)	Ozone formation (kg CFC-11 eq)	Abiotic depletion (kg Sb eq)
1	36.4	4,05E-04	2,03E-04	1,15E-09	9,94E-05
2	38.0	3,92E-04	2,02E-04	1,15E-09	9,94E-05
3	36.0	3,75E-04	1,96E-04	1,15E-09	9,94E-05
4	45.2	7,28E-04	3,63E-04	1,15E-09	9,94E-05
5	59.1	1,44E-03	6,24E-04	1,15E-09	9,94E-05
6	46.0	3,18E-04	1,86E-04	1,15E-09	9,94E-05
7	36.3	3,12E-04	1,82E-04	1,15E-09	9,94E-05
8	35.4	3,62E-04	1,94E-04	1,15E-09	9,94E-05
9	54.1	1,18E-03	5,43E-04	1,15E-09	9,94E-05
10	54.0	1,90E-03	5,42E-04	1,15E-09	9,94E-05
Regional average	43.9	8,42E-04	3,38E-04	1,15E-09	9,94E-05

Table 7. Environmental performance of 1 MJ of biodiesel from oilseed rape with optimal practices in Lorraine

Fertilization practice type	GWP (100 years) (g CO_2 eq)	Acidification (kg SO_2 eq)	Eutrophication (kg PO_4 eq)	Ozone formation (kg CFC-11 eq)	Abiotic depletion (kg Sb eq)
1	31.5	2,47E-04	1,66E-04	1,15E-09	9,94E-05
2	36.9	4,22E-04	2,05E-04	1,15E-09	9,94E-05
3	32.1	2,64E-04	1,70E-04	1,15E-09	9,94E-05
4	38.7	5,37E-04	3,09E-05	1,15E-09	9,94E-05
5	57.0	1,80E-03	5,56E-04	1,15E-09	9,94E-05
6	29.2	1,61E-04	1,49E-04	1,15E-09	9,94E-05
7	35.4	2,32E-04	1,76E-04	1,15E-09	9,94E-05
8	28.7	1,67E-04	1,50E-04	1,15E-09	9,94E-05
9	46.4	9,40E-04	4,78E-04	1,15E-09	9,94E-05
10	47.0	9,57E-04	4,82E-04	1,15E-09	9,94E-05
Regional average	37.6	5,44E-04	2,92E-04	1,15E-09	9,94E-05

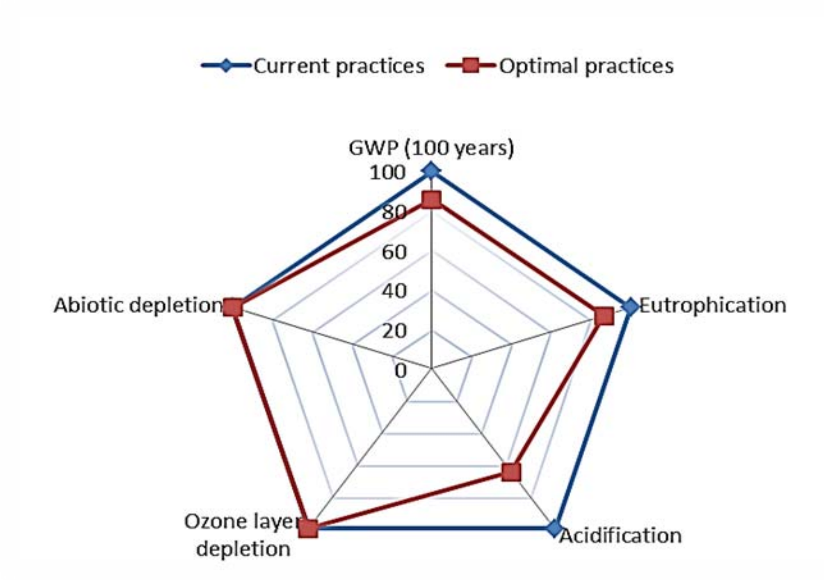

Figure 1. Impacts of fertilization practices improvement on LCA results of biodiesel from oilseed rape

4. Discussion

4.1. Reliability of simulated yields and nitrogen emissions

The modeling approach used here with CERES-EGC to estimate oilseed rape yields and Nr emissions related to its production takes into account the variability of local conditions and nitrogen fertilization practices adopted by farmers in the Lorraine region. However, since the model was run only for the year 2012 because of data availability, the effect of interannual climate variability was not tackled in this study. Thus, it would be desirable to run simulations for a longer duration and thus make it possible to place oilseed rape in a representative crop rotation of the region. This would probably significantly affect the modeled outputs.

In this study, we chose to accumulate simulated fluxes from the harvest of the previous crop to the harvest of oilseed rape because of lack of meteorological data availability. We thus attributed the emissions associated to the residue management of the previous crop. However some studies (eg, Gabrielle et al. 2014) opted for simulations from the sowing of the considered crop to the sowing of the following crop.

Direct N_2O emissions simulated by CERES-EGC were consistently lower than the estimates based on the Tier 1 IPCC guidelines. Regarding indirect N_2O emissions, the opposite trend was found. Thus, the total N_2O emissions simulated by CERES-EGC are quite similar to those obtainable with the IPCC methodology. By contrast, when comparing regional emissions related to current fertilization practices to those related to optimal practices, the difference between the two methodologies becomes significant. The decline in direct N_2O emissions after the adoption of optimal practices simulated with CERES-EGC is 5 fold lower than the decline calculated by using IPCC methodology. Thus, using IPCC guidelines would result a greater benefit of best management practices on the environmental balance of biodiesel from oilseed rape.

This study shows that practices based on organic fertilizers are causing significant emissions of N_2O compared to practices with only inorganic fertilizers. Because organic nitrogen remains available in the soil for the following crops and contribute to their yields, allocating part of the emissions occurring in the year following application to these following crops could be considered.

The simulated oilseed rape yields during the year 2012 were slightly overestimated. This could impact the environmental balance of biodiesel since using the observed yields instead would result in higher emissions per ton of grains harvested. However, the simulated yields were overall very similar to the oilseed rape yield recorded during the past five years, on average. They may thus be considered more representatives of yields for the area.

4.2. LCA outputs

With the current fertilization practices, the GWP intensity of biodiesel from oilseed rape obtained in this study is quite similar to that estimated by the European Commission for the whole Europe (i.e. 44 g CO_2 eq per MJ) , which used the same method of co-product handling. However, our estimated environmental performance of biodiesel from oilseed rape in Lorraine is most likely not representative of the whole France or Europe because of the specific characteristics of this region, in which yields are also usually below the national average.

In addition to the effect of yields and nitrogen emissions on the outputs of the LCA, the chosen allocation method has a key role in the environmental performance of biodiesel from oilseed rape. For example, the use of a mass-based allocation could reduce life-cycle GHG emissions of the biodiesel chain. The opposite could be observed if co-products were handled by system expansion. However when using this method, it should be borne in mind that some unrealistic assumptions which simplify market mechanisms are inevitable.

The modifications occurred in the inventory data related to feedstock production when we tested the effects of optimal fertilization practices are the source of the environmental balance of biodiesel improvement. Improving the environmental performance of the biorefinery would further reduce the LCA indicators of biodiesel.

5. Conclusion

The environmental impacts of biodiesel from oilseed rape were assessed in the French Lorraine region. The use of an agro-ecosystem model allowed considering local conditions and fertilization practices heterogeneity when estimating oilseed rape productivity and related reactive nitrogen emissions. LCA results showed that the GHG saving currently permitted by the substitution of fossil diesel is not consistent with the sustainability criteria to be respected at the European scale. Famers must opt for optimal nitrogen fertilization practices in order to mitigate the environmental impacts of the whole biodiesel chain. Such improvement in the crop management can lead to produce a more environmental friendly biodiesel, at least when land use change effects are neglected (both direct and indirect). However, these effects may be large and are currently coming in sharp focus from a policy and sustainability perspective.

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Assessing the materiality of various sustainability issues in the agrifood sector with LCA-based tools: 3 case studies

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ABSTRACT

The Global Reporting Initiative's (GRI) recently launched fourth generation guidelines (G4) strongly emphasising the concept of materiality as a key concept for choosing where an organization should put its focus within its sustainability-related activities. What *are* those issue areas in which an organization has its most significant economic, environmental and social impacts? How can they be identified and prioritized? Environmental life cycle assessment (E-LCA) and Social life cycle assessment (S-LCA) are timely and useful solutions to operationalize the Materiality principle in the agri-food industry. Conducted together, E and S-LCA allow for identifying significant social and environmental impacts of an organisation's activities throughout its products' life cycles, pinpointing the Social and Environmental Material Aspects. These tools can complement other methods used to assess materiality, such as stakeholder surveys, by adding additional objective and scientific verification.

Keywords: Global Reporting Initiative (GRI), Materiality principle, Material Aspects, Aspects Boundary, Reporting, Social and Environmental LCA

1. Introduction: The new GRI G4

The Global Reporting Initiative (GRI) is an organisation promoting "sustainability reporting as a way for organizations to become more sustainable and contribute to sustainable development" (globalreporting.org). GRI is mostly known for the reporting framework it developed: a holistic set of economic, social and environmental indicators. Reporting organisations are invited to present data covering GRI indicators and also information about business partners within reach of their sphere of influence, which can extend more or less within the value chain.

Until the recent release of the fourth version of the Global Reporting Initiative (GRI) guidelines (G4), the concept of the sphere of influence was leading sustainability report's boundary setting. Therefore, if the reporting organisation had the known ability to influence the practices of some of its business partners, on its own, these were reported on.

With G4 the focus has shifted from the concept of sphere of influence to materiality. In an effort to help companies assess and report on "what really matters", GRI is encouraging organisations to think beyond their fenceline and analyse the perceived and assessed significant economic, social and environmental impacts that their activities have on the planet, society and individuals.

Materiality is a concept stemming from accounting and being defined differently by standards such as SASB, IIRC and GRI. In this article, we will stick to the definition offered by GRI. G4 defines Material Aspects as issues "that reflect the organization's significant economic, environmental and social impacts" and incentivizes reporting organisations to only provide information on those aspects. Therefore GRI simultaneously try to limit the Aspects reported on and enlarge the scope of the value chain for which reporting is desirable. Organisations are encouraged to conduct a "materiality assessment" to identify the significant sustainability impact and the value chain links they should be focussing on.

Conducting a value-chain materiality assessment to understand where the biggest impacts occur, regardless of whether those impacts are within direct control, may be one of the greatest benefits and potential challenge for companies. Whether companies have a good grasp on their value chain sustainability impacts or need to build this exercise into their strategy, the new requirement will undoubtedly be a step forward helping companies to understand the bigger picture of sustainability performance across all their activities.

2. The importance of materiality for agri-food sector organization's sustainability strategy

Used in financial circles for decades, materiality assessment identifies issues that are critical to an organization's success, typically because of their implications to the company's financial situation or due to a high level of interest from their stakeholders. Outcomes can be used to direct business strategy, including fiscal investment, toward the most strategic areas.

The GRI's understanding of material aspects similarly narrows the universe of issues that a company reports on to those most critical to both the company and its stakeholders. The most significant change in materiality is the new consideration of the boundary attributes when determining material issues. This means that companies must not only consider what, but where an issue is relevant across the organization and its value chain (which sites, subsidiaries, countries, suppliers, products, etc.). It also means that a company may report a different boundary for different issues. For example, child labor could only be reported on from the perspective of the supply chain or specific buying categories or geographies in the supply chain, while greenhouse gas emissions could be reported on from the perspective of the company-owned fleet or the downstream impacts associated with product use.

A big potential gain with the increased importance and new approach to determining materiality is a more targeted and meaningful identification of relevant issues. One potential concern around materiality is that companies become too selective, screening out issues that they should be reporting. Deciding on a realistic list of material issues will be a critical element of the G4 reporting process.

This is especially relevant for agri-food sector companies characterised by complex and often overlapping supply chains with low transparency and numerous interacting environmental and social issue areas. While many agri-food companies have a good grip on Life Cycle Assessment, CERES and Sustainanalytics (2014) evaluation of the 24 companies of the CERES Gaining Ground report shows that while food and beverage companies have improved their sustainability performance overall, a more focused look at how they are addressing agricultural sourcing highlights the need for more action by the sector. In particular, it appears clearly that companies are just getting started. For example, almost half of the companies (11 of 24) do not disclose any evidence of conducting risk assessments of their agricultural suppliers.

Among CERES and Sustainanalytics key findings, current goals and commitments are often narrowly focused and targeted primarily on agricultural inputs, such as coffee, soy, and palm oil, which are subject to third-party verification schemes. Managerial oversight is weak with the vast majority of companies not adequately disclosing how these issues are managed internally. According to CERES, this raises concerns about the priority placed on addressing agricultural risks to the business.

In addition and across the board, companies are failing to effectively measure and disclose how their efforts are addressing sustainability risks, such as increased exposure to the adverse effects of climate change, and ultimately changing farmer practices. In contrast to the lack of risk assessment practices, 75 percent (18 of 24 companies) outline at least a basic strategy to mitigate one or more supply chain risks. In other words, rather than developing a corporate-wide approach that is based on a broad assessment of supply chain risk, most companies appear to addressing these issues on an ad hoc or project-specific basis.

CERES concludes that risk assessments are a critical component of prioritization, and companies should look to increase disclosure of these efforts to give stakeholders a better understanding of how the company plans to have impact. Tellus and Sustainanalytics (2012) also highlighted the lack of sustainability and especially social sustainability disclosure of agri-food companies. In the same line, OXFAM (2013) recently urged agri-food companies to foster more transparency, implement sustainability policies and codes of conduct effectively, and report more thoroughly on their sustainability impacts.

GRI G4 offers the right incentive for agri-food sector's companies to integrate more effectively their sustainability efforts.

3. The role of LCA in assessing materiality

What does a materiality assessment for the sustainability space look like? It's flexible, and there's not yet a clear precedent or procedure. Many companies are therefore struggling with questions of how to conduct such an assessment and how to know that a given approach has been thorough enough. To best characterize the set of issues that are material to a company, it makes sense to combine a science-based, objective approach offered through tools like life cycle assessment (LCA) with a more qualitative and subjective approach that can better account for stakeholder perceptions.

For the assessment of stakeholder perceptions, input from internal and external stakeholders is required. This might come in the form of interviews, surveys or both. A frequent option is to ask a set of stakeholders to rank issues in order of their importance to the company and in terms of the importance to the stakeholders and then to see where the issues fall in these rankings.

Crosswalk the importance of issues to internal stakeholders with their importance to external stakeholders to identify sustainability topics that rise to the top in every conversation. Combining stakeholder dialogue with quantitative tools can provide a robust approach for identifying material sustainability issues.

Although qualitative assessment of stakeholder input is useful, used alone it suffers from a potential for "group think", where all stakeholders rank highly the things that they hear and talk about frequently without objective checks of whether these are really the key issues. A more quantitative, science-based approach can be used to validate and/or provide some corrections to the list of issues identified by stakeholders. Many companies—through their work on product LCA and corporate footprinting—already have available some tools to do this.

Environmental LCA, social LCA and life cycle costing (LCC) enable us to compare the materiality of issues on an objective and quantitative basis. While stakeholder outreach identifies importance based on stakeholders' perspectives, LCA quantifies importance from the vantage points of the environment and society. Coupling these tools provides an organization with a robust approach to identifying its material issues.

How can we answer these questions with LCA tools? An LCA of a company's key products or a multi-indicator corporate footprint can identify the relative importance of issues. In fact, answering such questions is exactly the purpose of the endpoint metrics available in the most commonly used LCA impact assessment tools. An environmental LCA might indicate, for example, that the largest impact a given company has on the environment is in the areas of climate change, water use and toxic substance emissions, while issues such as ozone depleting substances, acidifying emissions, land use and smog-forming emissions are of much lesser importance. Regardless of whether these outcomes reinforce or contradict the input from stakeholders, they provide a valuable additional source of input to the materiality assessment process.

A multi-criteria footprint can tell us about the relative importance of various issues of concern. We can compare the categories at the "endpoint" to identify which are most impacting. This example justifies a heavy focus on a few issues.

The same now holds true for social issues. A report underscoring the importance of a life cycle-based approach to understanding and managing social risk in support of policies and decision-making was recently published by the European Union Joint Recent Center. The report, conducted a macro-scale analysis of the social risk profile of EU-27 imports by combining trade statistics regarding imports from intra- and extra-territorial trading partners in 2010 with country and sector-specific social risk indicator data in five thematic areas: Labour Rights and Decent Work; Health and Safety; Human Rights; Governance; and Community Infrastructure. It compared the apparent social risk profiles of EU-27 imports based on consideration of country/sector-of-origin social risk data only, compared to a life cycle-based social risk assessment which took into account the distribution of social risk along product supply chains.

The report shows that (1) the majority of social risks associated with imports to EU-27 countries are attributable to extra-territorial rather than intra-territorial imports, and (2) the risks of Injuries and Fatalities make the largest proportionate contribution to an overall, single-score measure of risk. However, these two approaches provide otherwise dissimilar "signals" as to the magnitude and distribution of social risk. The former approach would invariably prioritize interventions targeting only those direct trading partners known to have high levels of social risk in the sectors providing exports to EU-27 Member States. In contrast, the latter approach provides insight as to the distribution of risk along supply chains, which may be low in the sector of a given country exporting products to Europe, but high overall for those products due to the social risks associated with the activities that support production in that sector. Such supporting activities include physical flows of inputs such as raw materials and energy, and also the activities of service sectors. The life cycle approach hence affords a much more nuanced consideration as to for whom, where, and to what extent social risk may be of particular concern.

The study was conducted using the Social Hotspots Database (www.socialhotspot.org), a repository of social indicator data covering 225 countries and 57 economic sectors and relevant to five overarching thematic areas:

Labour Rights and Decent Work; Health and Safety; Human Rights; Governance; and Community Infrastructure. These indicators were developed based on the recommendations of the UNEP/SETAC Guidelines for Social Life Cycle Assessment (UNEP/SETAC, 2009), the ISO 26000 Guidelines for Social Responsibility (ISO, 2010), the Global Reporting Initiative (GRI) G3 Guidelines, (GRI, 2006) and the Global Social Compliance Programme (GSCP) Reference tools (GSCP, 2012) and are making use of over 200 sources of data. The Social Hotspots database also include a Global Input-Output model based on the Global Trade Analysis Project general economic equilibrium model.

The report demonstrates that it is now feasible for organizations to implement a life cycle approach to assess not only environmental impacts, but also social footprints, and that doing so will provides crucial insights regarding the materiality sustainability issues and this especially in economic sectors such as agri-food.

This example from the EU also helps to demonstrate that life cycle-based materiality assessment can quantitatively assist in identifying material impacts, issues, and risks which are themselves best measured in qualitative or judgemental ways. And this is both true and relevant for environmental as well as social issues of concern. While traditional LCA works with data reflecting units of "elementary flow" per unit of process output (e.g., kg of CO2 released to air or liters of water consumed, per unit of process output), this approach can be complimented by Life Cycle Attribute Assessment (Norris 2006).

Some issues of concern are better characterized through a qualitative or a summary expert judgment assessment about the overall attributes of processes. For example, forestry operations may be certified as sustainable or unsustainable, based on extensive, expert-based, site-specific assessments which take all sorts of context-relevant nuances into account in a systematic and pre-determined way. These assessments can provide *more* information than a purely quantitative characterization of the same forestry operation. Thus, a company or its stakeholders may want to assess to what degree the company's supply chain is sourcing sustainably-sourced forest product output, and to push for and document continuous expansion of such sourcing. Conversely, the company or stakeholders may identify as material any major instances of non-certified forest sector output in the supply chain.

Life Cycle Attribute Assessment (LCAA) enables users to harness quantitative models of product supply chains together with a mix of quantitative and qualitative information about the processes in those supply chains in order to identify material risks and opportunities for sustainability improvement. LCAA provides information about the *share of relevant activity* within supply chains that has any attributes of interest or concern to the company and its stakeholders.

In the case of the EU social supply chain assessment, the results highlighted the share of worker-hours which were at elevated risk relative to a comprehensive set of social indicators. And in the case of sustainable forestry assessment, LCAA can identify the share of forest acreage at elevated risk of unsustainable management – pinpointing where such management is occur geographically, where within the supply chain, and at what quantitative level of participation across the supply chain. Specific to the Agri-food sector, an LCAA could identify the share and major instances of land use which occur in high-risk zones for biodiversity, or the share and instances of crop cultivation which is certified organic, as two examples. Materiality can also arise via the interaction of two or more attributes, such as being located in region with endangered species and lacking certification of management practices which respect or enhance survivability of the relevant species.

This however shows that contrarily to financial materiality assessment where standardised revenue-related quantitative cut-off criteria exist to determine whether an item is material or not (a 10% threshold is suggested by many audit manuals), LCA-based materiality assessment for CSR reporting also needs to take into account subjectivity and judgment in the process. Stakeholders' perception, business strategy, geopolitical context, etc. all provide relevant yet qualitative information which must be addressed in the analysis.

4. Applying LCA-based materiality assessment in the Agri-food sector

As we stated in the introduction many agri-food companies and associations are already experienced applying LCA based tools. However, as we highlighted, frequently LCAs are conducted on a project basis and not as part of a corporate-wide approach. We selected three case studies to demonstrate how LCA can be mobilized in a wider strategic context.

4.1. The case of a leading coffee roaster

A coffee roasting company has an interest to understand the materiality of various social and environmental issues to support its strategy for measuring and disclosing its performance in the sustainability area to its business customers, investors and others. In addition to a questionnaire and set of interviews conducted with stakeholders, a review has also been done of several existing environmental LCAs of coffee systems to understand: a) among which environmental impact categories do the most important environmental impacts occur; and b) within which stages of the coffee life cycle are these impacts occurring.

Applying an endpoint impact assessment methodology allows the results to speak directly to the first of these two issues, providing a science-based weighting of more than a dozen mid-point impact categories based on the extent to which they cause impairments on ecosystem quality, human health and resource availability. In this case, it is found that carbon footprint is a highly important issue for both human health and ecosystems, land use is an additional important issues for ecosystems, and fossil fuel consumption is most important for resource availability. Many other environmental issues, such as ozone depletion, acidification and others are shown to be of much lesser importance. It is further acknowledged that because none of the references reviewed allowed for the inclusion of water quantity issues in the endpoint analysis, the review does not adequately address this issue (although this could be done with an LCA that does address this issue at the endpoint).

Regarding stages of the coffee life cycle, the work shows that for each of the key impact categories identified, the growing of coffee beans is a highly important stage. This stage is where the great majority of the land use impact occurs. It also corresponds to a significant area of focus for climate change. The brewing of coffee is a significant area of impact of fossil fuel consumption and for climate change. In addition, the cleaning of coffee brewing equipment and/or cups is identified as a potentially important area of focus, although this will vary widely depending on consumption and washing habits.

The value of this work is that it provides a science-based confirmation of several issues that have been identified by the company and its stakeholders as important, as well as providing additional justification for avoiding spending time on several of the issues that are shown to be of lesser importance.

4.2. The case of Pet Food

Several Pet food companies were interested to jointly identify their most significant social hotspots as a key input to the development of key performance indicators. The social hotspots database was used to provide a scoping assessment completed by a literature review and a stakeholder survey.

The study aimed at providing an overarching perspective on pet food social impacts. It was first necessary to identify the relevant sector, food products nec, from a specified list of 57 sectors used in the SHDB. This sector list corresponds to the sectors defined by the Global Trade Analysis Project's (GTAP) economic equilibrium model, which is used to develop the SHDB Worker Hours Model.

Next data on the location of the last manufacturing activities for pet food products sold in the US had to be found. The United States International Trade Commission's Interactive Tariff and Trade DataWeb indicated China, Canada, Thailand and the US to be pet food products sources for the US market. These countries, food products nec sector were tested using the SHDB to assess their supply chain impacts. The results were prioritized based on labour intensity, risk level and overlap.

The results showed that respectively, China, India, Thailand, Cambodia, Vietnam, Mexico and the US were countries responsible for a significant share of the total social impacts. The economic sectors of fisheries, food products nec, vegetables, fruits and nuts, were identified most often as being at high risk overall.

Labor rights and decent work, health and safety and to a lesser extent human rights were the most prominent types of impact identified. Low wages, migrant labour risks and fatal and non-fatal injuries were the most prominent contributing issues.

The results of the scoping assessment provided a necessary overview of some of the most salient social risks attributed to the pet food sector for products sold in the US. In particular the results served as a starting point to literature review and were a key consideration in the development of key performance indicators along with the stakeholder survey results.

5. Using LCA results in reporting – GRI and SAFA

Several Canadian Agri-food companies associations (for instance Canadian Sphagnum Peatmoss Association and the Fédération des producteurs de porcs du Québec) were interested to gain a better understanding of their environmental and socioeconomic performance. In these cases, the LCAs culminated years of efforts to document each association's sustainability profile by building on previous initiatives. In order to structure and organize in a coherent and synthetic way the results of these efforts but also to enhance and ease the communication of the work done so far, these two organisations commissioned CSR reports based on the FAO's SAFA framework (to be published in Spring 2014).

The SAFA Guidelines have been developed to "provide an international reference tool for assessing the sustainability performance of food and agriculture enterprises" [1] all across a product' supply chain, from production to retailing. To do so, SAFA draws upon several internationally recognised reference tools, including the ISO norms for Life Cycle Assessment and the GRI Sustainability Reporting Guidelines.

Integrating LCA results to the SAFA reporting framework was an opportunity for these organisations to focus on those issues that were identified as social and environmental "hotspots" and to define and communicate objectives directly related to them. For instance, the Quebec Pork Producers Federation decided to adapt the reporting framework – originally built on 4 dimensions (Governance; Economic resilience; Environmental integrity; Social well-being) – to add "Animal welfare" as a core aspect of social responsibility within the industry as the S-LCA demonstrated. On the environmental side, the Canadian Sphagnum Peatmoss Association has reported in more details and has adopted more ambitious commitments related to the sector's water and carbon footprints since these issues were identified as important "hotspots" within the sphere of influence of the companies, i.e. on which the sector could directly act. In these two cases, the LCA result provided sound and credible data that have legitimated the organizations, both internally and externally, in the choice of information to report and in the nature of the engagements to adopt.

6. Conclusion

GRI G4 positions materiality assessment as a crucial component of sustainability reporting. Since G4 also expands the scope of reporting to supply chains issues, it makes materiality assessment even more relevant. G4 is not the only Social Responsibility instrument to invite companies to conduct supply chains risk assessments. The UN Business and Human Rights framework also encourages companies to conduct comprehensive supply chain due diligence.

Life Cycle Assessment is a technique which provides science-based information related to the potential sustainability impacts of companies, products and services. It can be used on a project basis but it can also be used to develop or support a comprehensive sustainability strategy. It supports materiality assessment by providing an objective overview of supply chains impacts, their relative contributions to the overall risk, and which phases and locations appear to be most significant.

LCAs supplement stakeholder surveys by providing comprehensive risk information that can include issues or phases of the supply chain which may otherwise be overlooked; at the same time, it may illuminate the small relative impact of issues or activities which were perceived as being more critical in the absence of a holistic and systemic assessment. It provides companies additional support in prioritizing the key activities and issues that truly drive their sustainability impacts and performance..

Two of our examples demonstrated how Agri-food sector companies are using LCAs to identify and prioritize impacts as part of reporting and KPI development processes. Our third example presented how this information helps companies to then report using sector specific reporting guidelines such as SAFA.

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Environmental impacts of genetic improvement in growth rate and feed conversion in fish farming under density and nitrogen limitation

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ABSTRACT

Many environmental impacts can be attributed to fish farming and there is a need to explore new ways of reducing environmental impacts, such as fish genetic improvement. The environmental consequences of genetic improvement in thermal growth coefficient (TGC) and in feed conversion ratio (FCR) were investigated, therefore, in an African catfish farm in Recirculating Aquaculture System. A Life Cycle Assessment study was conducted and combined with a bioeconomic model of genetic response. Results show that environmental consequences of genetic improvement in TGC and FCR are different among impact categories, depending on their main contributors and on the factor limiting production at farm level. Genetic improvement of FCR always leads to lower environmental impacts while improving TGC decreases environmental impacts only when density was the limiting factor. Those results are important for the future development of selective breeding programs in fish farming taking into account environmental impacts.

Keywords: genetic improvement, life cycle assessment, fish farming, feed conversion ratio, thermal growth coefficient

1. Introduction

Fish farming can have many environmental impacts, such as emission of pollutants (Read and Fernandes, 2003), use of natural resources (Naylor et al., 2000) and eutrophication (Folke et al., 1994). Using life cycle assessment (LCA), feed production and farm operations were shown to be the main stages contributing to environmental impacts of fish farming (Aubin et al., 2006; Pelletier et al., 2009). The environmental impacts of alternative feed compositions or alternative rearing systems were, therefore, investigated. Some studies explored the potential of plant-based diets or organic feed to reduce impacts (Boissy et al., 2011; Papatryphon et al., 2004; Pelletier and Tyedmers, 2007), whereas other studies explored the potential of a recirculating aquaculture system to reduce impacts (Aubin et al., 2009; d'Orbcastel et al., 2009). These studies, however, clarified existing trade-offs between different environmental impacts when changing feed composition or rearing conditions. It appeared difficult, therefore, to decrease all environmental impacts of fish farming. Consequently, there is a need to explore new ways of reducing environmental impacts of fish farming systems.

Genetic improvement, realised through selective breeding programs, is expected to change environmental impacts of livestock production by increasing productivity or improving production efficiency (Wall et al., 2010). The magnitude and the direction of this change in fish farming is, however, not known. Wall et al. (2010) suggested to model emissions at farm level in order to determine the environmental consequences (or environmental values) of a change in traits in order to evaluate the capacity of each trait to decrease environmental impacts. This approach is similar to the framework used to calculate the economic value of economic important traits (Groen, 1988).

We explored, therefore, the capacity of a genetic improvement in thermal growth coefficient (TGC) and in feed conversion ratio (FCR) to decrease environmental impacts of 1 t of African catfish produced in a recirculating aquaculture system (RAS). We chose to investigate TGC and FCR for two reasons. First, fish breeders consider TGC as the main trait to improve in order to increase farm profitability. FCR is also an important trait because the majority of fish reared in intensive systems are carnivorous species, which require high amounts of protein and lipid in their diet, making fish diets expensive. Second, we investigated TGC and FCR because genetic improvement of these two traits is susceptible to affect environmental impacts through higher productivity and better production efficiency. A RAS was studied for three reasons: (1) it is a highly controlled system, which is easy to model, (2) it has a potential as a development model for fish farming because of better waste and water management and (3) the existence of technical limiting factors for production level is well known. In practice, limiting factor can influenced farm management and breeding goals have to be adapted for such limiting factor (Gibson, 1989; Groen, 1989). The factors limiting production are the maximum nitrogen treatment capacity of the bio-filter and the maximum fish density in rearing tanks. Changing TGC and FCR is expected to change the limiting factor and the limiting factor might influence environmental values of both traits.

The capacity of a genetic improvement in TGC and in FCR to decrease environmental impacts was investigated through environmental values calculated using a bioeconomic model combined with a LCA.

2. Methods

2.1. System boundaries

A cradle-to-farm-gate analysis was applied including three stages: feed production, farm operation and waste water treatment (figure 1). The farm studied was a typical commercial Dutch farm producing 500t of African catfish per year in an indoor RAS. Fish processing, hatchery operations as well as sludge agricultural utilization were not taking into account.

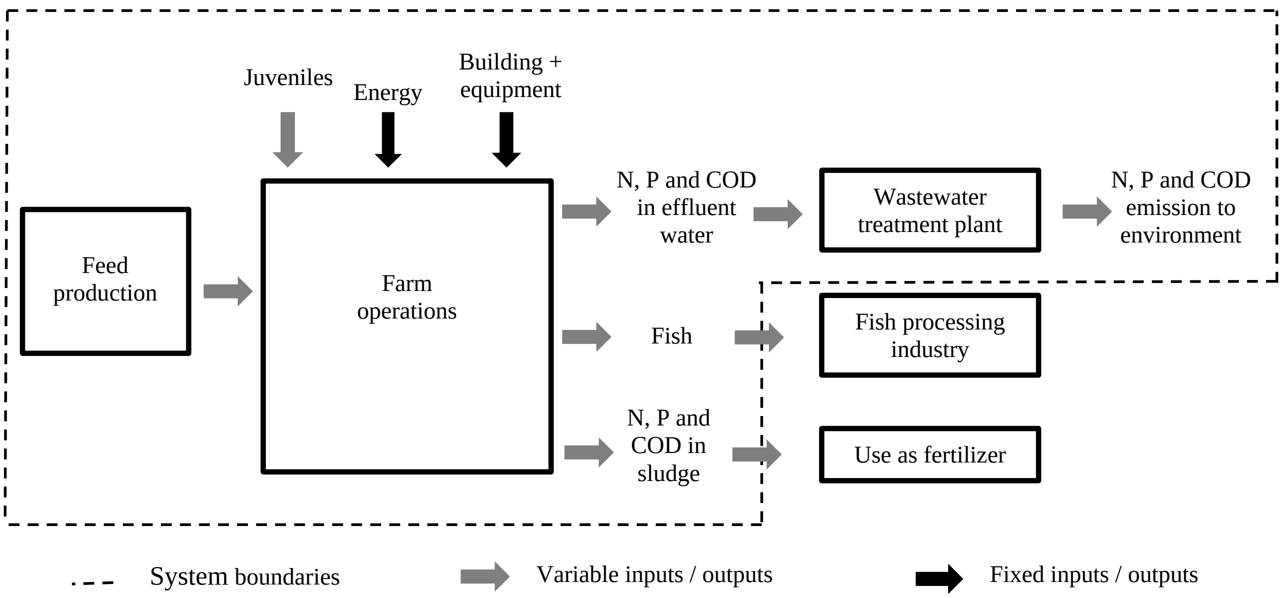

Figure 1. Diagram of the system studied.

2.2. Life Cycle Inventory

Feed production. Crops-derived ingredients were coming from Brazil and France. Fish-derived ingredients were coming from Peruvian fish milling industry and from by-products of Norwegian fish milling industry. In case of a process yielding multiple products, such as soy bean oil and meal, the environmental impact of the process was allocated to the multiple outputs based on their relative economic value (i.e. economic allocation). The transport of feed ingredients to feed manufacture in France was assumed to be by transoceanic ship and by lorry (>32t). The resources required to produce the feed were found in the literature (Pelletier et al., 2009). The transport of pellets from France to the fish farm in Eindhoven were assumed to be by lorry (>32t).

Farm operations. We investigated a farm producing 500t of African catfish per year in an indoor RAS. In this indoor system, water was thermo-regulated around 27 °C through regulating the ambient air. The RAS was composed of four main parts. (1) Rearing tanks growing fish from 13g to 1300g. The maximum density in the tank was 230 kg/m^3, which was one of the limiting factors. (2) A mechanical filter which removed solid waste.

(3) A bio-filter processing nitrification with a nitrification rate of 100%. The nitrification capacity of the bio-filter was limited to 40 kg of dissolved NH_3-N per day, which was the second limiting factor. (4) A denitrification reactor using manure of the fish as C-source with a denitrification rate of 80%.

The amount of equipment used (i.e. pump, tanks), the surface of building needed and the consumption of energy at farm level were considered fixed. The emission of nutrients, the consumption of feed and the production of fish were, however, dependent on growth rate and feed efficiency, which could affect the level of production. In this study, growth rate was expressed through Thermal Growth Coefficient (TGC) (Dumas et al., 2007) and feed conversion was expressed through Feed Conversion Ratio (FCR). A bioeconomic model was developed to calculate annual fish production, annual feed consumption at farm level and annual emissions of nutrients after waste water treatment according to TGC and FCR values. The individual emission of nitrogen (N), phosphorus (P) and chemical oxygen demand (COD) were calculated using a mass-balance approach (Cho and Kaushik, 1990). The proportion of dissolved and solid fractions emitted by the fish was estimated from the digestibility of feed components. Retention capacity of the drum filter, nitrification capacity of the bio-filter and denitrification capacity of the denitrification reactor were used to calculate emission of nutrients in effluent water and in sludge.

Waste water treatment. Effluent water, highly concentrated in nutrients, coming from the fish farm was disposed in a waste water treatment plant. We considered a typical waste water treatment plant running in Europe, including three treatment stages: mechanical treatment, biological treatment, chemical treatment. It also included sludge digestion via fermentation. The treatment capacity of the waste water treatment was used in order to evaluate the amount of nitrogen, COD and phosphorus released into water. 28% of the COD, 75% of the nitrogen and 52% of the phosphorus coming from the farm were, therefore, released into water. Life cycle inventory of water treatment as well as the secondary data were extracted from ecoinvent v2.2 database (Swiss Centre for Life Cycle Inventories, 2010).

2.3. Life cycle impact assessment

The environmental impact of the analysed system was related to a functional unit of 1 t of fish. Four environmental categories were investigated: eutrophication, acidification, climate change and cumulative energy demand (CML2 method and Simapro software) and expressed per t of fish. Land use change, however, was not taking into account. The analysed system was divided in the following sub-systems: (1) feeds consumption, including processing, production of feed ingredients and transportation; (2) N, P and COD emission from biological transformation of the feed at farm level and after waste water treatment; (3) effluent water treatment at wastewater treatment plant; (4) equipment use, including its manufacturing, transport and use; (5) buildings use, including material production and transportation, construction and use; (6) energy consumption (electricity, gasoline, heat gas), including production, transportation and use. Environmental impacts of each of these sub-systems were calculated separately.

Then, the results of these environmental impacts were multiplied by the quantity required at farm level calculated from the bioeconomic model. Feed consumption and nutrients emission are variable at farm level depending on TGC and FCR values. Environmental impacts at farm level of the two sub-systems feed production and nutrients emission are, therefore, variable. In contrary, use of building, use of materials and energy consumption at farm level are considered fixed. Environmental impacts at farm level of those three sub-systems are, therefore, fixed.

Environmental impacts at farm level were divided by annual fish production to expressed environmental impacts per ton of fish. Environmental impacts were calculated for four TGC values corresponding to three generation of selection from the current population mean with a percentage of genetic improvement of 6.8 as calculated by (Sae-Lim et al., 2012). TGC values tested were: 8.33, 8.93, 9.53 and 10.11. For FCR, a wide range of values was tested from current population mean FCR = 0.81, which is a value commonly observed in African catfish farms, to FCR = 0.64.

2.4. Environmental values

The environmental values (ENV) of the four impact categories (eutrophication = E_V, acidification = A_V, climate change = CC_V and cumulative energy demand = CED_V) of a trait t {FCR,TGC} express the environmental impact, in percentage, of a unit change in one trait while keeping the other trait constant. ENV of both traits were calculated in three steps:

1) Calculate environmental impact per ton of fish (i.e. $A_{\mu t}$) using current generation means for trait t (μ_t). The current generation mean (or reference scenario) is 8.33 for TGC and 0.81 for FCR. The TGC of the current population is the result of 119 days required for the fish to grow from 13 to 1300 g with a daily average temperature of 27°C. FCR was set at 0.81 in order to balance cost with revenue when TGC = 8.33.

2) The mean of trait t is increased by Δ_t while keeping the mean of the other traits constant. $\Delta_{TGC} = \mu_{TGC} \times 6.8\%$ and $\Delta_{FCR} = \mu_{FCR} \times -7.6\%$ as calculated by (Sae-Lim et al., 2012). The next generation means are, for instance, TGC = 8.93; FCR = 0.81 and TGC = 8.33; FCR = 0.75. The model was run a second time to calculate environmental impacts.

3) ENV were calculated for trait t as:

$$A\_V_t = \frac{(\mathbf{A}_{\mu t+\Delta t} - \mathbf{A}_{\mu t})}{\mathbf{A}_{\mu t}} \quad \text{Eq. 1}$$

Environmental values were calculated for 2 situations where the combination of TGC and FCR was: TGC = 8.93; FCR = 0.81 and TGC = 8.93; FCR = 0.69. These two situations were chosen because the limiting factor was different and also because one generation of selection in both trait did not change the limiting factor.

3. Results

Table 1. Percentage of contribution of the different sub-systems to the four impact categories in the reference scenario where TGC = 8.33 and FCR = 0.81.

	Feed production	Nutrients emission	Effluent water treatment	Building use	Material use	Energy use
Acidification	57.5%	0%	0.3%	10.2%	27.3%	4.7%
Eutrophication	33.6%	62.2%	0.2%	0.1%	0.3%	3.1%
Climate change	72.3%	0%	0.8%	1.5%	3.9%	21.4%
Cumulative energy demand	68.7%	0%	0.68%	2%	4.9%	23.6%

Table 1 shows the contribution of each different sub-system to the four impact categories in the reference scenario where TGC = 8.33 and FCR = 0.81. The impact of nutrient emission on acidification is 0% because effluent water are directly directed to effluent water treatment plant, which is responsible of 0.3% of the total acidification. In table 1, we can see that the main contributors to the four impact categories are different. For acidification, climate change and cumulative energy demand the main contributor is feed production (respectively 57.5%, 72.3%, 68.7%). The second contributor to those three impact categories are fixed sub-systems at farm level, building use for acidification (27.3%) and energy use for climate change (21.4%) and for cumulative energy demand (23.6%). In opposite, the two main contributors of eutrophication are nutrients emission (62.2%) and feed production (33.6%).

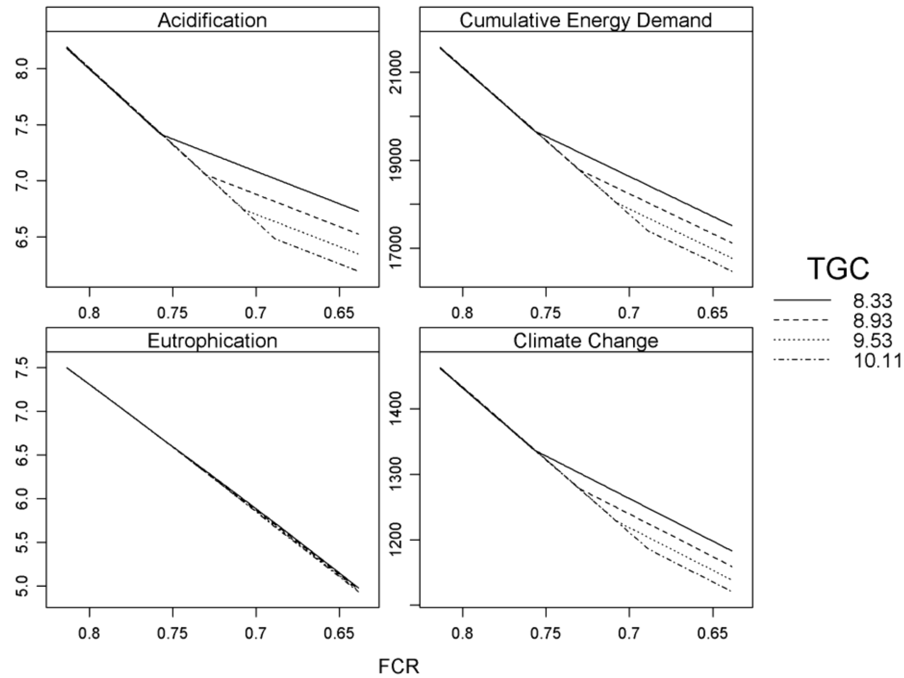

Figure 1. Environmental impacts calculated per t of fish for four impact categories as a function of FCR for four values of TGC.

Figure 1 shows the results of environmental impacts for the four impact categories investigated. First of all, the combination of TGC and FCR defines whether dissolved NH_3-N or density is the limiting factor. Environmental response to genetic improvement in TGC and FCR are different among impact categories according to the factor limiting production. These differences are because TGC and FCR have different effects on production. Two effects are capable of decreasing environmental impacts per unit of fish produced: increasing productivity (annual fish production) and increasing production efficiency (feed consumed per ton of fish). Increasing productivity, while keeping the same production efficiency, dilutes fixed environmental impacts at farm level, such as building use, over more fish produced. Increasing production efficiency, while keeping the same productivity, decreases the amount of feed required to produce one ton of fish and decreases the amount of nutrients emitted per ton of fish which decreases environmental impacts. Depending on the limiting factor, improving TGC or FCR changes either productivity, production efficiency or both.

Dissolved NH_3-N as limiting factor. For instance, when TGC is 8.33 and when FCR decreases from 0.81 to 0.75, the factor limiting production is dissolved NH_3-N. In this situation, improving FCR decreases environmental impacts of the four impact categories while improving TGC does not influence it.

Improving FCR decreases environmental impacts because individual daily excretion of nutrients decreases, which allows to stock more fish per batch to reach dissolved NH_3-N limitation. Consequently, when FCR is improved, both productivity and production efficiency increases, which decreases environmental impacts per ton of fish produced (figure 1).

Additionally, improving TGC does not influence environmental impacts because individual daily excretion of nutrients increases, which constrains to stock fewer fish per batch. This lower number of fish per batch is offset by rearing more batches per year. Consequently, when TGC is improved, productivity and production efficiency remain constant, which keep the environmental impacts at the same level (figure 1).

Density as limiting factor. For instance, when TGC is 8.33 and FCR is lower than 0.75, the factor limiting production is the density of fish in rearing tank. In this situation, the number of fish harvested per batch if fixed.

Improving FCR, therefore, decreases feed consumption per ton of fish and decreases nutrients emission per ton of fish. Consequently, when FCR is improved, only production efficiency increases, which decreases environmental impacts per ton of fish produced but with a lower rate than in dissolved NH_3-N limitation situation (figure 1).

Additionally, improving TGC increases the number of batch harvested per year, which increases annual production of fish. Consequently, when TGC is improved, only productivity increases, which decreases environmental impacts (figure 1).

Different environmental responses to genetic improvement in TGC can, however, be observed when density is the limiting factor. Improving TGC decreases acidification, climate change and cumulative energy demand while the response in eutrophication is smaller. The response to genetic improvement in TGC for eutrophication is smaller because improving TGC increases productivity only, which dilute fixed environmental impacts at farm level, such as use of material, over more fish produced. The contribution of fixed environmental impacts to eutrophication is, however, smaller than the contribution of fixed environmental impacts to acidification, climate change and cumulative energy demand (table 1).

Table 2 gives ENV_{TGC} and ENV_{FCR} of the four impacts categories investigated in both situations, when dissolved NH_3-N is the limiting factor and when density is the limiting factor.

When dissolved NH_3-N is the limiting factor, all ENV_{TGC} are null as TGC do not alter environmental impacts. These results can be explained by the fact that increasing TGC does not influence productivity nor production efficiency. The ENV_{FCR} are, however, negative because when dissolved NH_3-N is the limiting factor, increasing FCR increasing productivity and production efficiency, which decreases environmental impacts.

When density is the limiting factor, ENV_{TGC} have a negative value because increasing TGC increases productivity which dilutes fixed environmental impacts at farm level over more fish produced. E_V_{TGC} is, however, very close to zero because eutrophication is relatively insensitive to fixed environmental impacts at farm level compare to other impacts categories (table 1). The ENV_{FCR} are also negative but CED_V_{FCR}, E_V_{FCR} and A_V_{FCR} are lower when density is the limiting factor than when dissolved NH_3-N is the limiting factor because increasing FCR when density is the limiting factor increases production efficiency only. Moreover, E_V_{FCR} is higher than CED_V_{FCR}, E_V_{FCR} and A_V_{FCR} because the two main contributors of eutrophication are nutrients emission and feed production, which are sensitive to production efficiency.

Table 2. Environmental values of TGC (ENV_{TGC}) and FCR (ENV_{FCR}) of four impacts categories (eutrophication = E_V, acidification = A_V, climate change = CC_V and cumulative energy demand = CED_V) calculated when dissolved NH_3-N is the limiting factor (TGC = 8.93, FCR = 0.81) and when density is the limiting factor (TGC = 8.93, FCR = 0.64).

ENV	Trait	TGC = 8.93 FCR = 0.81 Limiting factor = dissolved NH_3-N	TGC = 8.93 FCR = 0.69 Limiting factor = density
A_V	FCR	-10.3%	-4.3%
	TGC	0%	-2.7%
E_V	FCR	-11.8%	-13.4%
	TGC	0%	-0.3%
CC_V	FCR	-9.3%	-5.5%
	TGC	0%	-1.8%
CED_V	FCR	-9.6%	-5.2%
	TGC	0%	-2.0%

4. Discussion

The results of our simulations show that environmental response to genetic improvement in TGC and FCR is different when density and when NH_3-N are the limiting factors. When NH_3-N is the limiting factor, only genetic improvement in FCR decreases environmental impacts because it increases productivity and production efficiency. On the contrary, when density is the limiting factor both genetic improvement in TGC and FCR decreases environmental impacts because improvement in FCR increases production efficiency and improvement in TGC increases productivity. When density is the limiting factor, however, environmental response to genetic improvement in TGC and FCR are different across impact categories depending on their main contributors. Eutrophication is mainly driven by feed production and nutrients emission, which are influenced by production efficiency. Conversely, other impact categories are mainly driven by feed production and building use or energy use, which are influenced by productivity.

d'Orbcastel et al. (2009) investigated the impact of a RAS producing rainbow trout using two different values of FCR, 1.1 and 0.8. This range corresponds to 27.3% of improvement, or to 3.6 generations of selection when percentage of improvement in FCR is 7.6%. They found, therefore, that decreasing FCR by 7.6% decreased acidification by 5.8%, eutrophication by 4.3%, climate change by 6% and cumulative energy demand by 2.4%. It is, however, difficult to use the results from d'Orbcastel et al. (2009) as a comparison basis for our results because they didn't take into account changes that could occur in farm management when FCR changes. d'Orbcastel et al. (2009) considered that improvement in FCR increased production efficiency but, using dynamic modelling of the relationship between genetic improvement and farm management, our results show that improving FCR can also increase productivity when NH_3-N is the limiting factor. Changes in FCR can also lead to changes in limiting factor.

The results presented here confirm that FCR would be the major trait to include in the breeding goals for decreasing environmental impacts. In fish breeding, however, FCR is a difficult trait to improve as it is difficult to measure individual feed intake. FCR is expected to be correlated to TGC, however, studies diverge on this subject. In rainbow trout, Kause et al. (2006) predicted that selection only for daily gain increases daily gain by 17.6% per generation and simultaneously increases feed efficiency (1/FCR) by 8.4%. In parallel, some other studies in salmonids did not observe any correlation between growth rate and feed efficiency and showed that genetic gain in growth is due to higher feed intake, while feed efficiency remain unchanged (Mambrini et al., 2004; Sanchez et al., 2001).

5. Conclusion

Environmental values could be included in breeding programs in a similar way as economic values and could help fish breeders to design effective breeding programs in terms of environmental consideration by putting more emphasis on the relevant traits, in a specific limiting factor situation. Our results have, therefore, important implications for fish breeders who may need to alter their breeding objectives depending on what is limiting production on the farms of their customers. We show that environmental values of FCR and TGC are dependent on the factor limiting production. Improvement of feed efficiency always improves environmental impacts in any situation. However, selecting for increased growth rate is only relevant in situations where nitrogen emissions are not limiting production. Those results are important for the future development of selective breeding programs in fish farming taking into account environmental objectives.

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LCA of perennial crops: implications of modeling choices through two contrasted case studies

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ABSTRACT

As highlighted in several recent reviews, there is a need to harmonize the way LCA of perennial crops is conducted (Bessou et al. 2013; Cerutti et al. 2013). In most published LCA on perennial crops, the agricultural production is based on data sets for just one productive year. This may be misleading since performances and impacts of the system may greatly vary year by year and the evolution of the stand over the cycle induces specific mechanisms (nutrient re-mobilization, yield alternating, resistance etc.) that must be included. Without a proper mechanistic model, the only way to account for such phenomena is to wider the data sets to at least account for each stage of the stand development and, if possible, all years of the crop cycle. Three modeling choices for the perennial crop cycle were tested in parallel in two contrasted LCA case studies: oil palm fruits from Indonesia, and small citrus from Morocco. Modeling choices tested were: i) a chronological modeling over the complete crop cycle of orchards (Bessou et al. 2013), ii) a three years average from the productive phase and iii) a selection of different single years from the productive phase. In both case studies, the system boundary included all processes from the seed production until the harvested fruits at farm-gate. The functional unit was 1 kg of fresh fruits. The chosen approach to model the perennial cycle influences the final results and deserves specific attention.

Keywords: perennial crop, LCA, chronological modeling,

1. Introduction

First LCAs including perennial crops were mostly cradle-to-grave assessments of renewable products or services based on agricultural feedstock, e.g. biofuel, heat and power, biomaterial. Despite the importance of the cropping system in terms of impact contribution over the whole process chain (e.g. 10%-80% of the total primary energy input and more than 25% of the GHG balance in most bioenergy chain *In* JEC, 2008), cultivation remained secondarily addressed (Fazio and Barbanti 2014; Monti et al. 2009). Hence, in most published LCAs on perennial crops, data on the cropping system is scarce and is often based on one productive year without accounting for the whole perennial production cycle (Bessou et al. 2013). However, assessing perennial crops in the same way as annual ones may induce bias notably due to the variability in practices and yields over the plantation lifespan or the potential importance of changes in carbon stocks (Mithraratne et al. 2008). Recent reviews highlighted the need to better account for the specificities of perennial cropping systems within LCA and to harmonize the way LCA of perennial crops is conducted (Bessou et al. 2013; Cerutti et al. 2013; Cerutti et al. 2011).

The aim of this study is to illustrate the potential bias due to varying choices in modeling the perennial crop cycle within cradle-to-gate LCAs. The baseline assumption is that a chronological assessment is the closest way to model the real perennial crop cycle, since it uses a continuous data set for a single plantation plot followed over its whole lifespan (Bessou et al. 2013). We hence compared this chronological assessment with two other ways to model the cycle: 1) a three-year average of three consecutive productive years; and 2) various single years taken randomly within the productive phase, which may be seen as approximating perennial to annual crop. We selected two contrasted case studies in different pedoclimatic conditions and with different crop managements in order to test the sensitivity of the modeling choices to contrasted perennial cropping systems. The first case study concerns oil palm fruit production in Indonesia; the second one implies small citrus produced in Morocco. We first present the data sets used for the LCI and the LCIA results, then we discuss the effect of modeling choices and further needed improvements to increase the accuracy of perennial crop LCAs.

2. Methods

2.1. LCA goal and scope

The LCA goal is to assess the environmental impacts of the production of 1 kg fresh fruit from continued agricultural land use. The approach is an attributional LCA without any assessment of rebound effect. There is hence no consideration of any direct or indirect land use change. The system boundary is a cradle-to-farm gate one (Figure 1), including land preparation and all upstream processes to produce, transport and apply inputs to the field, plus field operations and field emissions up to fruit collection at the edge of the orchard or plantation. Downstream processes to conserve or transport the fruits to storehouses are excluded. The land preparation consists of the destruction of the previous crop (as for a re-planting procedure), hole digging, and seedling planting. Seed and seedling productions are accounted for as field inputs. Seed and seedling productions, as well as land preparation, are amortized over the whole crop cycle. At the field gate, there is no co-product production; crop residues are recycled internally in the field. Allocations or system expansion were not needed in these case studies.

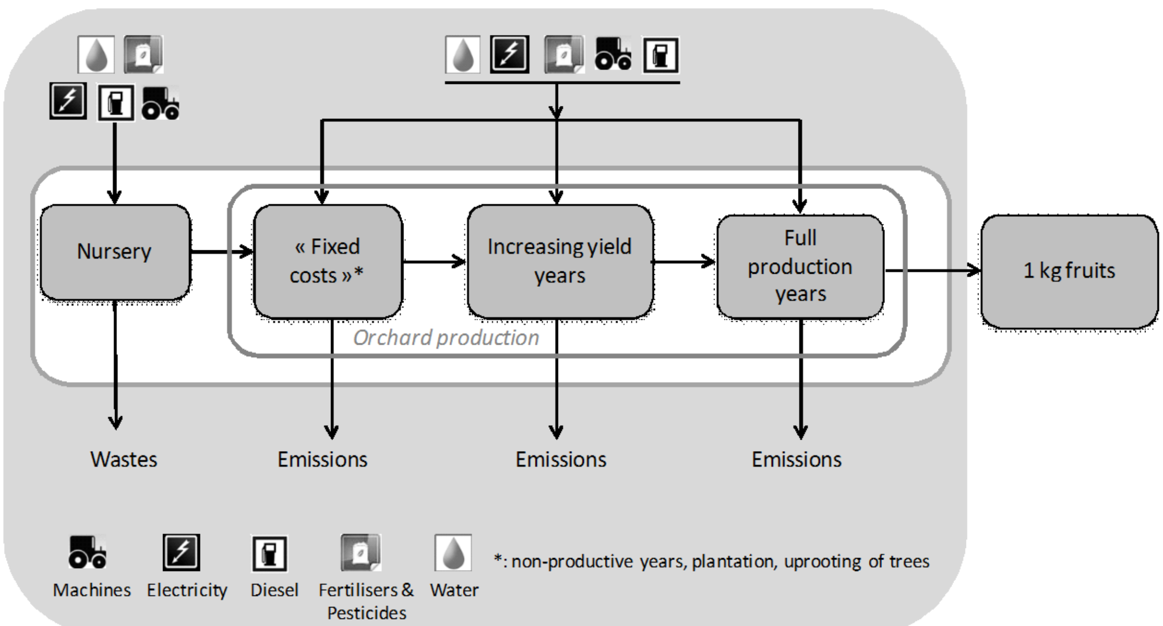

Figure 1. Simplified flow diagram for studied systems for oil palm from Indonesia and small citrus from Morocco

2.2. Data source and LCI

Primary data were collected through field surveys (2010-2013) and are recorded within the CIRAD LCA DATABASE ©-2014. These encompass the following parameters characterizing the cropping systems: input origins, types, doses, and transport distances; fuel consumption for all field operations; planting density and yield outputs. Details are given for each case study in sections 2.2.1.-2.2.2. Secondary data were taken from Ecoinvent v.2.2 database and encompass: production of synthetic inputs (fertilizers, pesticides, material to the nursery) and energy vectors (fuels, electricity); machines and fuel consumptions for input transportation (oversea and road transports); machines for field operations.

2.2.1. Oil palm fruit system

The studied oil palm production system consisted of an industrial plantation block on mineral soil (Typic Dystropept, Acrisol) in Riau Province, Sumatra, Indonesia. Data on seed and seedling productions were based on both company standards and field interviews. Data on land preparation were also based on company standard. The palms were planted in 1992 with a density of 136 trees/ha. The plantation was not irrigated, and water use only took place at seed and seedling production stages as well as for pesticide dilution. Primary data on fertilizer inputs were collected over the period 1992-2012. Yield records started after the immature phase and covered the period 1995-2012. Finally, pesticides and field operations were recorded over the period 2008-2012 (Table 1). Fertilizers were applied mechanically twice a year, whereas pesticide application and harvest were done manually.

Table 1. Key agronomic data for the oil palm fruit system in Indonesia; [min and max values].

	Units	Average for the non-productive years (0 - 2 years)	Average for the productive years (3 - 21 years)	Average for the whole cycle	Average for the last three years of the cycle	Year 1995	Year 1996	Year 2006
Fertilizers								
N	kg/ha/yr	38 [32 ; 44]	112 [109 ; 156]*	101	109	156	109	109
P_2O_5	kg/ha/yr	44 [38 ; 51]	16 [15 ; 31]	20	15	31	15	15
K_2O	kg/ha/yr	86 [13; 193]	43 [41 ; 82]	49	41	82	41	41
MgO	kg/ha/yr	7 [6; 8]	32 [0 ; 46]	28	37	46	27.5	37
Borate (boric acid)	kg/ha/yr	3 [0; 4]	4 [0 ; 6.5]	4	6.5	0	0	6.5
Fossil fuel	L/ha/yr	4.3	4.3	4.3	4.3	4.3	4.3	4.3
Total pesticides	kg/ha/yr	0.03	0.03	0.03	0.03	0.03	0.03	0.03
	L/ha/yr	0.84	0.84	0.84	0.84	0.84	0.84	0.84
Yield	t/ha/yr	0	22 [10.9 ; 30.5]	19	22	11	25	21

* Minimum and maximum values encountered in specific years over the time period of the productive years. Treatments during the non-productive years do not vary annually. Pesticides are herbicides used on a routine basis to clean circles and harvest pathways.

2.2.2. Small citrus system

In agreement with our Moroccan partner: a big producer of agricultural products, a 9-years old small citrus orchard was selected from the region of Beni Mellal in Morocco. This orchard showed recent technologies of production and had already reached its full production phase. The variety of small citrus was “Sidi Aïssa” grafted on “Citrange Troyer”. Over the first 9 years, detailed accounting data were collected to describe all agricultural operations. Large variations of input rates and yield were observed across the first nine years. The orchard was assumed to last 25 years. From the 10th to the 25th years an annual agronomical scenario was designed with the partner based on average data for all inputs and yield for the years 7, 8 and 9. The average yield of 42 t/ha corresponded to the expected yield by the farmer. Key agronomic data for the different phases of the small citrus orchard are presented in Table 2. The orchard, planted at a density of 5 x 4 m was fertigated. Several pumps were used to pump water in the groundwater and lagoons allowed 10 days of autonomy in water during the dryer season in case of pumps’ failure.

Table 2. Key agronomic data for the three phases of a nine years old small citrus orchard in Morocco. Data for the full production phase are extrapolated from the average data for the 3 previous years (7-9). [min and max values]

	Units	Non-productive years (0 - 3 years)	Increasing yield years (3 - 9 years)	Full production years (9 - 25 years)
Fertilizers				
N	kg/ha/yr	55 [46 ; 69]	155 [66 ; 224.5]	214
P_2O_5	kg/ha/yr	24 [8.5 ; 43]	48 [24 ; 67]	65
K_2O	kg/ha/yr	1.6 [0 ; 4.6]	140 [57 ; 221]	186
Fe	kg/ha/yr	0.45 [0.44 ; 0.46]	0.4 [0.04 ; 1.5]	0.8 [0.17 ; 1.48]
Zn	kg/ha/yr	0	0.39 [0.12 ; 0.9]	0.43 [0.20 ; 0.64]
Mn	kg/ha/yr	0	0.51 [0.03 ; 1.31]	0.57 [0.21 ; 0.91]
Irrigation				
Water (groundwater)	m3/ha/yr	6112 [5835 ; 6496]	7982 [6633 ; 11054]	8906
Fossil energy	L/ha/yr	1305 [1158 ; 1428]	1550 [1091 ; 1723]	
Electrical energy	kWh/ha/yr	0	0	7661
Total insecticides	kg/ha/yr	0.52	3.35	4.79
Total herbicides	kg/ha/yr	2.58	1.92	2.25
Total pesticides	kg/ha/yr	3.10	5.27	7.04
Yield	t/ha/yr	0	29 [5.5 ; 66]	42

2.2.3. Field emissions

For oil palm system, field emissions were calculated as following. Emissions due to fertilizer field application were estimated according to IPCC 2006 Tier 1 guidelines for both N- and C-compounds. P-compounds emissions were estimated based on SALCA-P model (Prasuhn, 2006) considering only run-off and leaching risks, in our case studies of perennial plantation with permanent soil cover on zero-slope land area (0-3%). Finally, heavy metals and active substances from applied pesticides were assumed to all end up completely in the soil. Heavy metals contents were recorded for synthetic fertilizers (Freiermuth, 2006) and organic fertilizers (primary data from measured recycled crop residues).

For small citrus: Ammonia, NO_2, phosphate and pesticides emissions were estimated following recommendations from Nemecek and Kägi (2007). Nitrous oxide emissions were based on IPCC emission factors but following Brentrup et al. (2000) approach. Following Brentrup et al. (2000) again, the nitrate leached was evaluated by calculating the leachable nitrogen from a nitrogen budget and by applying a drainage factor based on a water budget and the field capacity of the soil. As part of the N budget, N export in fruits was based on Vannière (1992) and N sequestered in trees (roots, stand, branches) was modeled using expertise from H. Vannière.

2.3. Modeling of the perennial crop cycle

Three modeling choices for the perennial crop cycle were tested in parallel in the two contrasted LCA case studies. Modeling choices tested were: 1) a chronological modeling over the complete crop cycle of orchards, 2) a three years average from the full production phase and 3) a selection of different single years from the full production phase. The chronological assessment consists in describing the whole cycle following the historical course of the crop development (Bessou et al., 2013). This approach is the closest way to the reality of a perennial crop development, since delayed effects from agricultural practices or intrinsic crop physiology features can be accounted for. However, the data set on the whole cycle of a perennial crop is hardly available in most of the cases. Therefore, it is important to try to quantify the bias due to truncated perennial crop modeling.

When individual years or the 3-year average were considered in this study, inputs upstream the investigated years were not accounted for, *i.e.* land preparation and planting, seeds and seedling production and all other years, including the costs of the non-productive years. The comparison between individual years and the 3-year average aimed at assessing how much of the inter-annual variability could be captured and how far it might then a better proxy to model the whole cycle compared to individual years.

2.4. Impact characterization method

The impact assessment was performed using the ReCiPe Midpoint life cycle impact assessment method (Goedkoop et al., 2009), adopting the Hierarchist perspective. The following environmental impact categories were considered: climate change (100 years IPCC 2007; kg CO_2eq), terrestrial acidification (g SO_2eq), freshwater and marine eutrophication (g P-eq and g N-eq respectively, based on the nutrient-limiting factor of the aquatic environment), terrestrial and freshwater ecotoxicity (g 1,4-DB-eq: 1,4-dichlorobenzene), agricultural land occupation (m^2.year), water depletion (m^3-eq) and fossil depletion (kg oil-eq). The non-renewable energy consumption (fossil and nuclear; MJ-eq) was assessed using the Cumulative Energy Demand method (Hischier et al. 2009).

3. Results

Across the cradle-to-farm-gate life cycle of oil palm fruit (Figure 2a), the productive years that account for 18/21 of the whole cycle contributed to the larger share of most impact categories (75-95%) except for freshwater eutrophication and water depletion. Since the palm plantations were not irrigated, the water depletion category, which only considers used tap water, was half related to used water to produce seeds (germination cycles) and irrigate the seedlings in the nursery, and half due to pesticides dilution during both non-productive and productive years. Across the other impact categories, fertilizers (production, transport and field emissions) during productive years contributed greatly to climate change, terrestrial acidification, marine eutrophication, and fossil depletion (70-90%); whereas other interventions (including mostly pesticides application) contributed more to the toxicity impact categories (~75%) but also to freshwater eutrophication (25%), water depletion (28%), and fossil depletion including fuel use (22%). Fossil depletion is notably related to fossil fuel used for field operations such as annual plantation maintenance or fertilizer broadcasting.

Across the cradle-to-farm-gate life cycle of small citrus, the tree planting and non-productive years (non-productive stage) contributed between 6.5% for terrestrial acidification and 29% for terrestrial ecotoxicity (Figure 2b). For categories other than toxicity impacts and land occupation, fertilization and irrigation represented the two main contributors. This was due to the production of fertilizers and fossil fuel (for irrigation) and their emissions after use at field level. For terrestrial ecotoxicity, the on-field emission of pesticides was the almost exclusive contributor shared between the non-productive stage (29%) and the productive years (70%). The share of non-productive stage was large for terrestrial ecotoxicity due to the application of toxic insecticides at the nursery stage (abamectin) and during the three first years of trees (methomyl). The freshwater ecotoxicity was due primarily to irrigation (production and combustion of fossil fuel) and secondarily to field pesticide emissions for productive years (27.6%) and for non-productive years (18.8%). Water depletion was due almost exclusively to water use for irrigation shared between non-productive years (8.8%) and the rest of the orchard's life (91%). Finally, agricultural land occupation was mostly due to the productive years (87.4%) and secondarily to non-productive years (12%).

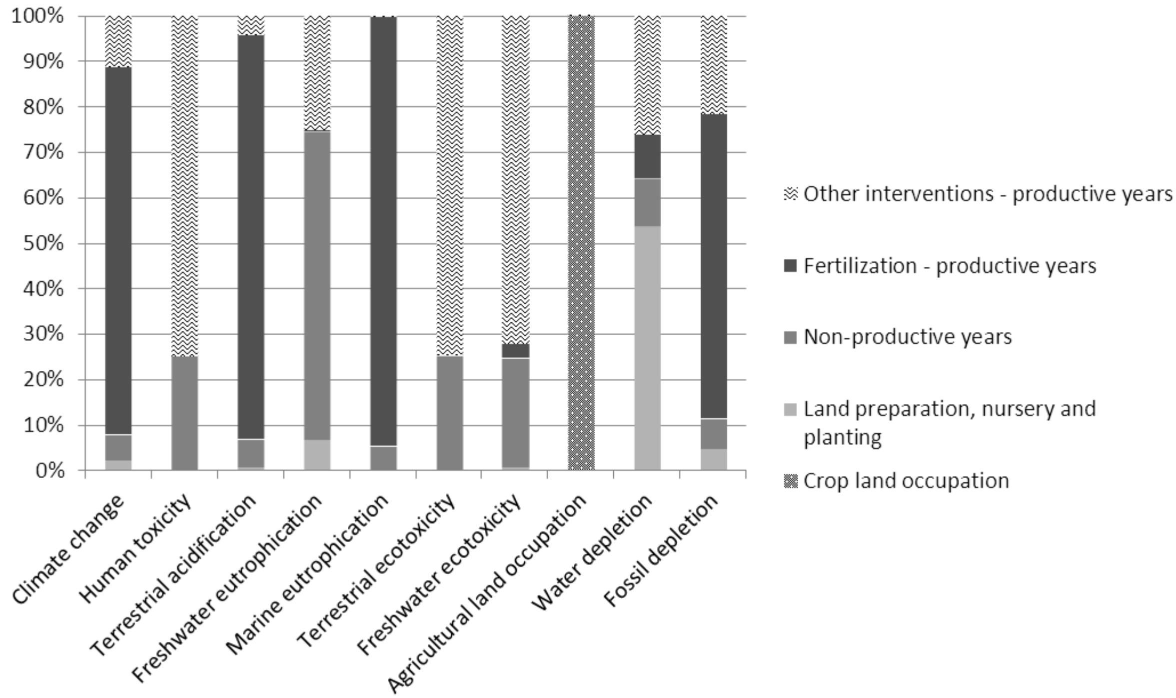

Figure 2a. Contribution analysis from cradle-to-farm-gate for environmental impacts (ReCiPe-Midpoint (H)) of palm oil fruit from Indonesia. Results are expressed per kg of fresh fruit bunches

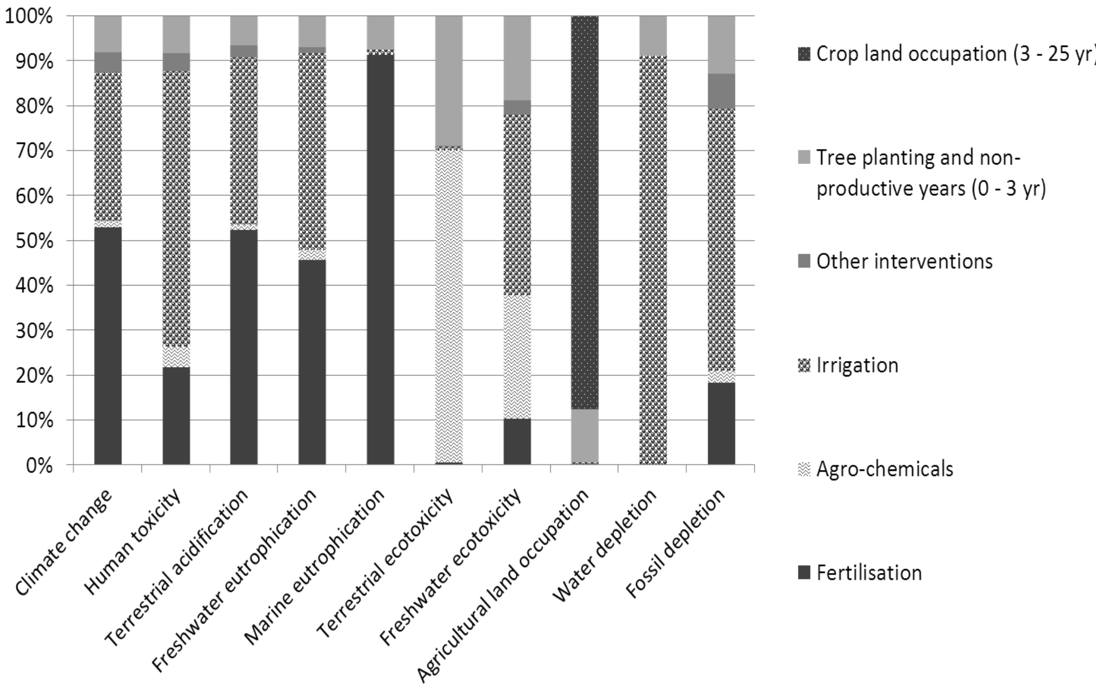

Figure 2b. Contribution analysis from cradle-to-farm-gate for environmental impacts (ReCiPe-Midpoint (H)) of small citrus from Morocco. Results are expressed per kg of raw fruit.

3.1. Sensitivity analysis to modeling approach for perennial crop cycle

In Figures 3a and 3b, modeling approaches are compared to the baseline scenario modelling the whole cycles. In both cases, the results were highly sensitive to the modeling approach used for the perennial crop cycle. Relative variations in results among the approaches are not homogeneous across impact categories.

In the case of oil palm fruit, the year 1995 showed a very specific pattern with lower yields and more inputs that the average productive years.

In the case of small citrus, compared to the full cycle scenario, the 3-years average scenario showed results between 47% for freshwater ecotoxicty and human toxicity up to 138% for marine eutrophication. As such, it did not appear as a good proxy for the full crop cycle because it excluded the non-productive years but also because it relied only on three years over 25 and did not reflect the whole orchard cycle properly. The single year scenarios showed extreme variations due to the yield variations, ranging from 20 t/ha for year 9 to 66 t/ha for year 8, but also to annual variations of rainfall, water use for irrigation and input rates. For instance, the year 9 scenario showed a very high marine eutrophication due to both a low yield and a high nitrate leaching due to a humid weather while year 7 and year 8 were very dry and associated to a nil nitrate leaching.

By selecting randomly one single year from the full production phase to evaluate a full crop cycle, the results could either be dramatically overestimated or underestimated. The most variable results were observed for the most climate-dependent impacts such as marine eutrophication and toxicity impacts due to variations in rainfall and pest pressure but also for toxicity impacts, to the use of different active molecules from one year to the other.

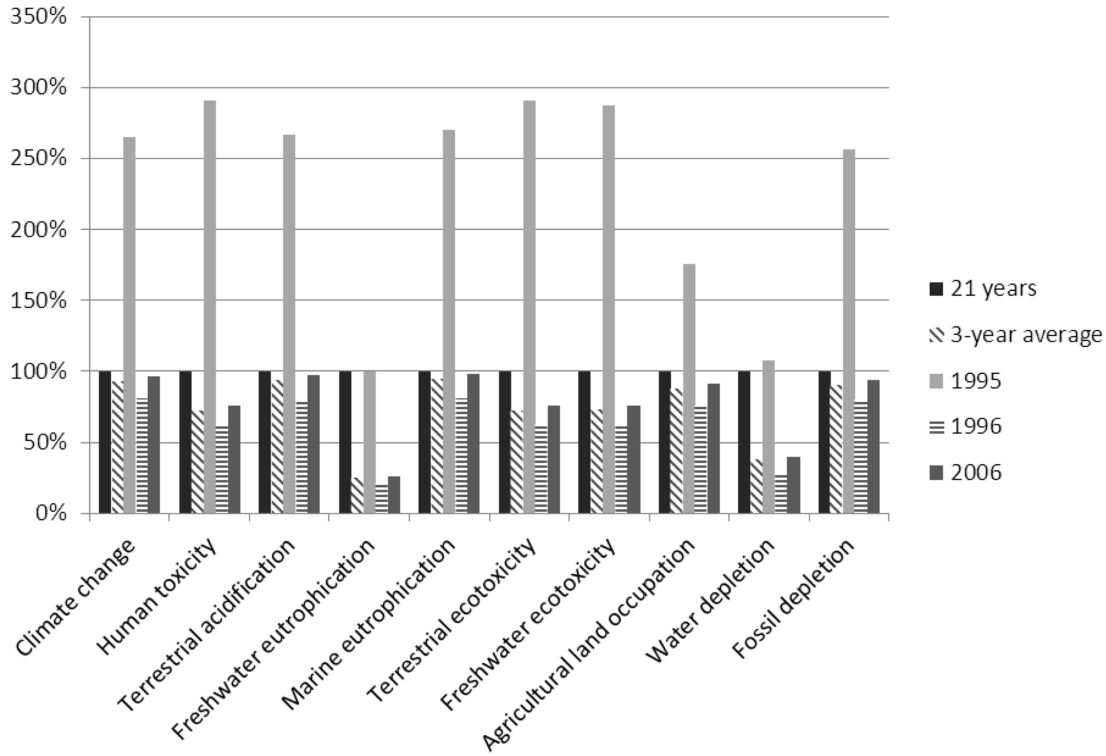

Figure 3a. Environmental impacts (ReCiPe-Midpoint (H)) for different modeling approaches of the perennial crop cycle for oil palm fruits from Indonesia: full cycle modeling (21 years, reference baseline 100%), 3-years average, years 7, 8 and 9. Results are compared per kg of fresh fruit bunches.

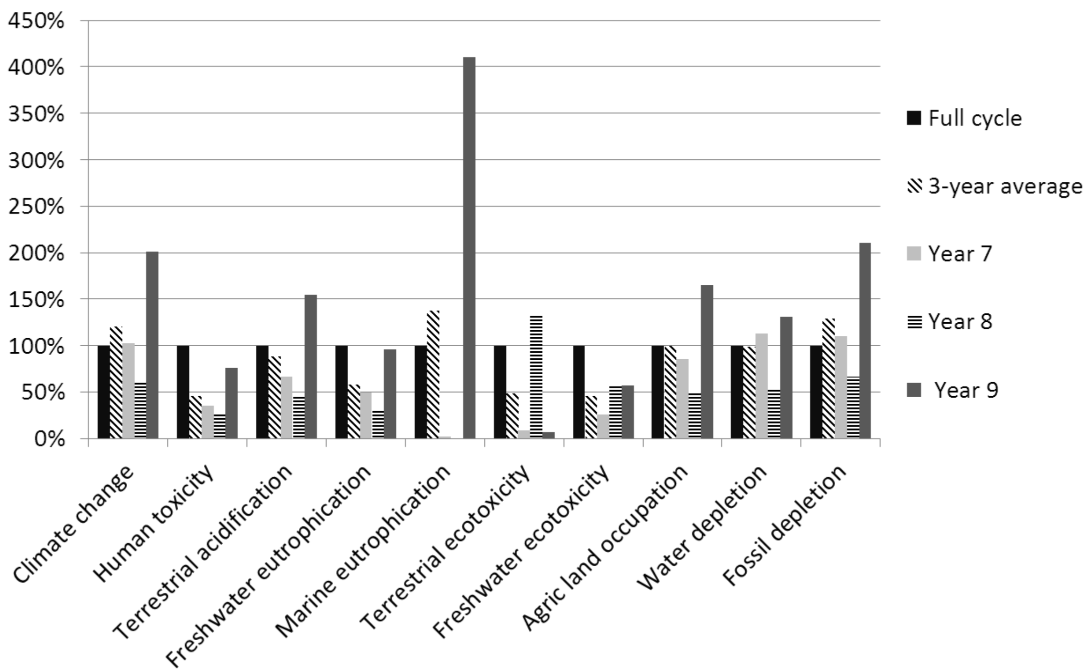

Figure 3b. Environmental impacts (ReCiPe-Midpoint (H)) for different modeling approaches of the perennial crop cycle for small citrus from Morocco: full cycle modeling, 3-years average, years 7, 8 and 9. Results are expressed per kg of raw fruit.

4. Discussion and conclusion

In two contrasted perennial case studies, one on palm oil from Indonesia, the other on small citrus from Morocco, different modeling approaches were developed to account for the perennial crop cycle. The baseline scenario included a complete modeling of the crop cycle while a 3-year average scenario and 3 single year scenarios were also tested. Key insights from these two analyses were consistent:

1. non-productive years have a large share in the environmental impacts of orchards and should be included;
2. choosing one single year from the full production phase leads to highly uncertain results and should be avoided especially for strongly alternating yield crops;
3. even a 3-year average scenario is not sufficient to capture properly the full perennial crop cycle and can be misleading
4. an effort should therefore be made to include the whole crop cycle ideally based on real data when available or at least on expert knowledge.

Although showing very different features, the two case studies contributed to draw consistent conclusions on the modeling of perennial crops in LCA. Analysing two contrasted LCA case studies, we highlighted the specific character of perennial crops in LCA and how important is the inclusion of all their phases in the assessment to account for their highly variable inputs and outputs over years. Other crucial and specific aspects of perennial crops especially in the Southern countries, which are under-represented in current statistical models to estimate field emissions (Bouwman et al. 2002; Stehfest and Bouwman 2006), still warrant further research and better modelling notably regarding:

(i) the inclusion of land use change (if any);
(ii) the modeling of field emissions of nutrients and pesticides combining parameters of soil, climate and practices and long-term recycling and re-mobilising mechanisms;
(iii) the inclusion of impacts due to water use.

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Introduction of grass-clover crops as biogas feedstock in cereal-dominated crop rotations. Part II: Effects on greenhouse gas emissions

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ABSTRACT
In an analysis of climate effects, increased soil organic carbon will have a dual effect due to both increased soil fertility and carbon sequestration. Even so, soil carbon changes are neglected in many crop production LCAs. In the present study, the introduction of grass-clover crops in cereal-dominated crop production was evaluated. The grass-clover crops were used for biogas production, and the digested residue was recycled to the farm as biofertilizer. A shift from the cereal-dominated crop rotation to integrated production of food crops and one or two years of grass-clover crops used as biogas feedstock would result in avoided emissions of 2-3 t CO_2-eq. ha^{-1} a^{-1}. Integrated food and energy crop production would in this case improve soil organic carbon content at the same time as resulting in considerably decreased greenhouse gas emissions from the cultivation system.

Keywords: soil organic carbon, energy crop, biogas, integrated production

1. Introduction

The use of crops or the removal of crop residues from farm land for biofuel production is often controversial, and has been identified as having negative impact on soil quality and food crop productivity (Lal 2010). The range of crops that can be used for biofuel production is, however, large, and it can be contra productive to generalize since the impact of crop choice on the cultivation system is complex and variable. An important aspect in the evaluation of changed land use is the impact on soil organic carbon (SOC) content. In an analysis of climate effects, increased SOC will have a dual effect due to increased soil fertility and crop yields and due to the carbon sequestration (Lal 2004). Even so, changes in SOC are neglected in many crop production life cycle assessments (LCAs) (Brandao et al. 2011). Loss of SOC, erosion and compaction are some of the degradation processes that are threatening soil fertility throughout the EU (EEA 2002; Soilservice 2012). Restoration of SOC content can be achieved by e.g. application of biofertilizers containing organic matter and by including green manuring, catch and cover crops in the crop rotation (FAO 2002). However, intensive agriculture is often niched to either crop production or animal husbandry. In Sweden, farms with animal husbandry cultivate grass-clover crops for forage, and have access to animal manure as biofertilizer, both which positively impacts soil SOC content (Jordbruksverket 2014). In regions with intensive food crop production on the other hand, there is no market for forage crops and the biofertilizer availability is low.

In the present study, which is presented in two parts, the purpose was to evaluate the introduction of grass-clover crops in a cereal-dominated crop rotation in a region with stockless farming and intensively cultivated clay rich soils. The calculations were performed as a farm based case study. At the case farm, the low input of carbon in the intensive food crop production the last decades has been identified as problematic with regards to the high clay content in the soils. Problems with soil compaction and crop losses due to standing water are common, and the grain yield is lower on the farm than in the region in general due to soil compaction in combination with low soil carbon content. Thus, the conventional food crop production maintained for decades was no longer seen as sustainable. The approach investigated in the present study was to integrate 1-2 years of grass-clover crops in the food crop rotation and use this as feedstock for biogas production. The purpose was to evaluate a scenario where a biogas plant has been integrated in an agricultural region with mainly stockless farming, where the biogas plant takes on the role of the absent ruminants. The presence of the biogas plant could create a market for the grass-clover forage as energy crop, and the biogas process also produces a digestate (the liquid residue after the biogas production), which can be used as biofertilizer. The overall objective of the project was to analyse how an integrated production of food crops and energy crops for biogas production impacts SOC and food crop production, which is presented in Part I of the study (Prade et al. 2014), and greenhouse gas emissions per land area, which is presented in the present paper, compared to the presently used food crop rotation.

2. Scenarios for crop production

The case farm (56°N, 12°E) includes 650 hectare (ha, 10 000 m^2) of medium to very heavy clay soils with soil clay content up to 65%, with a 4-year cereal based crop rotation typical for the region, including winter oil seed rape, winter wheat, winter wheat and oats. The reference scenario in the present study is based on data on yields measured at the farm for this crop rotation. In two scenarios, the crop rotation is extended to also include one year of grass-clover crops following oats (Scenario GC1), or two consecutive years of grass-clover crops following oats (Scenario GC2). This reduces the land used for production of food crops within the farm by 20% (GC1) and 33% (GC2).

The information on cultivation inputs for each investigated crop rotation is given as average per ha and year (a, annum) in Table 1. Operations include crop cultivation, fertilization with mineral fertilizer or biofertilizer, harvest, and for the biogas feedstock grass-clover also ensiling, transport to biogas plant and feed-in at biogas plant. Use of machinery and materials in different production operations was analyzed in order to calculate direct and indirect greenhouse gas (GHG) emissions, and include emissions from a large range of inputs as seeds, pesticides, concrete and plastics for ensiling etc and are summarized as corresponding GHG emissions as carbon dioxide equivalents (CO_2-eq) (Table 1). For details on these inputs, please refer to Björnsson et al. (2013). The main contributors to GHG emissions in cultivation (fertilizer production and diesel consumption) are given separately as amounts (kg fertilizer or liter of diesel) (Table 1). The fertilizer demand is the average demand of nitrogen (N), phosphorous (P) and potassium (K) in the crop rotations.

Table 1. Crop cultivation inputs

Scenario			Reference	GC1	GC2
		N	196	183	179
Crop fertilizer demand	[kg ha^{-1} a^{-1}]	P	28	27	28
		K	42	76	103
Diesel	[l ha^{-1} a^{-1}]		69	84	87
Materials	[kg ha^{-1} a^{-1}]	CO_2-eq	192	197	176
Machinery	[kg ha^{-1} a^{-1}]	CO_2-eq	92	107	94

Crop yield data and coefficients used for calculation of crop residues and SOC are presented in Part I of this study (Prade et al. 2014). Amounts of crop residues are calculated based on Nordic data in the base case, and in the sensitivity analysis the methodology presented by IPCC is applied (IPCC 2006). The amounts of crop residues are important for calculations of SOC, and in addition give rise to biogenic nitrous oxide (N_2O) emissions. The N content of the crop residues as presented by IPCC are used (IPCC 2006) for these calculations.

3. Biogas production

The problems with soil compaction and the declining grain yields at the case farm were the main reasons for constructing a biogas plant within the farm boundaries in 2006. Biogas is there produced from mainly food industrial waste. The digestate is used as biofertilizer on all land within the farm since 2007. Life cycle inventory data from a LCA performed for the biogas plant within the boundaries of the investigated case farm was used as input for the biogas part of the present study (Lantz and Börjesson 2014).

For the grass-clover as biogas feedstock, the emissions from cultivation inputs given in Table 1 include cultivation and harvest, field drying (to 35% dry matter, DM), transport to biogas plant, ensiling in bunker silos (assuming a DM loss of 5%) and feed-in at the biogas plant. The amounts of grass-clover silage available as biogas feedstock after losses are shown as average per ha of the crop rotation in Table 2. After feed-in, the grass-clover silage is assumed to be pretreated by extrusion to improve the properties as biogas feedstock, where after it is kept at 37°C under oxygen free conditions in the stirred tank biogas digester. The digestate is stored in a covered digestate storage tank and subsequently recycled to the farm as biofertilizer. Loading, transport to field and spreading of the digestate as biofertilizer is included in the cultivation inputs shown for GC1 and GC2 in Table 1. The biogas produced is upgraded, compressed, spiked with propane (a requirement to compensate for the lower energy value of the upgraded biogas compared to the gas in the Swedish natural gas grid) and delivered to the natural gas grid. The compression to 200 bar required at the vehicle filling station is also included.

GHG emissions were calculated based on mass of biogas feedstock, mass of digestate or the biogas produced for each scenario as presented in Table 2. Background information on emissions or primary energy demands are given as footnotes below the table. Emissions in biogas production are calculated to represent only the emissions related to the grass-clover, even if in practice this feedstock will be co-digested with other biogas feedstock. Similarly, the amount of biofertilizer in the form of digestate is calculated to correspond only to the product from grass-clover digestion. Thus, the emissions or benefits from digestate originating from other organic feedstock (food industrial waste etc.) are excluded.

Table 2 shows the amount of digestate and the content of N as total nitrogen (N-tot) and ammonium nitrogen (NH_4-N), P and K. These calculated concentrations are based on measured content in the grass-clover feedstock (Björnsson et al. 2013) and a calculation model for nitrogen mineralization during biogas production based on methane yields as presented by Lantz et al. (Lantz et al. submitted). This digestate is used as biofertilizer in the GC1 and GC2 scenarios, and then partly replaces the mineral fertilizers needed to fulfil the crop fertilizer demand (Table 1), while in the reference scenario, mineral fertilizers only are used.

Table 2. Input and outputs from biogas production in the grass-clover scenarios

Scenario			**GC1**	**GC2**
Amount biogas feedstock [a]	[t ha^{-1} a^{-1}]		5.9	10.5
Biogas production [b]	[GJ ha^{-1} a^{-1}]		17.3	30.9
Amount of digestate [c, d]	[t ha^{-1} a^{-1}]		4.8	8.5
Applied to field as digestate [e]	[kg ha^{-1} a^{-1}]	N-tot	54	87
		NH_4-N	28	42
		P	6	10
		K	32	52

[a] Primary energy input per t feedstock: electricity for extrusion 50 MJ t^{-1}, electricity for pumping and stirring 54 MJ t^{-1}, natural gas for process heat 121 MJ t^{-1} (Lantz and Börjesson 2014).

[b] Based on methane yields of 261 m^3 t^{-1} DM and 221 m^3 t^{-1} DM for 1st and 2nd cut grass-clover (Björnsson et al. 2013) and after methane losses in production (0.29%) and upgrading (1%) (Lantz and Börjesson 2014). Primary energy input in gas handling: upgrading 3.0% of energy in upgraded gas, compression 2.6% of energy in upgraded gas. Propane addition; energy corresponding to 25% of the energy in the upgraded biogas.

[c] Calculated as average for the crop rotation (as if the amount is distributed evenly spread evenly on the farm).

[d] Losses of nitrogen and organic material during digestate storage under roof cover are subtracted: 1% of total nitrogen (N-tot) is lost as ammonia nitrogen (NH_3-N). Methane emissions are calculated based on the IPCC model for manure (IPCC 2006) with a scenario specific calculated maximum methane potential (B_0) and a methane conversion factor of 3.5% (Naturvårdsverket 2013), resulting in methane emissions during digestate storage of 0.4% (GC1) and 0.5% (GC2) of the produced methane (Björnsson et al. 2013).

[e] Only the N present as NH_4-N is assumed to replace mineral N. P and K are assumed to replace mineral fertilizer without losses.

4. Method

The LCA was performed according to the ISO standard (ISO 2006), with focus on quantifying emissions of greenhouse gasses (GHG). The functional unit was set to 1 ha of arable land. The assessment included cultivation, harvest and storage of crops, biogas production, upgrading and compressing, digestate storage and application and soil carbon changes. The majority of the LCI data are summarized in Tables 1 and 2. Additional emission and characterization factors are summarized in Table 3. Data on harvest yields, SOC modelling and SOC changes are presented in Part I of the study (Prade et al. 2014). A systems expansion approach, in accordance with the recommendation in the ISO standard, was applied. In the systems expansion, the total output of grains (wheat and oats) and oil seed (rape seed) was equivalent in the different scenarios. Thus, a reduced output of grains and oil seeds per ha in the crop rotation as a whole, due to the introduction of grass-clover crop cultivation, was compensated for by additional grain and oil seed production outside the farm. The potential benefit of improved food crop yields due to SOC increases as outlined in Part I of this study was thus not taken into account here (Prade et al. 2014). The required additional cultivation was assumed to take place within the region on excess farmland, not leading to any indirect land use changes due to displacement effects. Regional data for GHG emissions in cultivation (Börjesson et al. 2010) were used, and recalculated per kg DM of crop grain or seed (Table 4). The output of upgraded biogas delivered to vehicle filling stations via the natural gas grid was assumed to replace fossil vehicle fuels (EU 2009) (Table 4).

Table 3. GHG emission and characterization factors

Characterisation factors in aggregation of emissions		Reference
CH_4	23 g CO_2-eq (g CH_4)$^{-1}$	(IPCC 2006)
N_2O	296 g CO_2-eq (g N_2O)$^{-1}$	(IPCC 2006)
Life cycle emissions from input energy or materials		
Diesel	84 g CO_2-eq MJ^{-1}	(EU 2009)
Swedish electricity mix [a]	10 g CO_2-eq MJ^{-1}	(Gode et al. 2011)
Nordic electricity mix [b]	69 g CO_2-eq MJ^{-1}	(Gode et al. 2011)
Wood chips [b]	2 g CO_2-eq MJ^{-1}	(Gode et al. 2011)
Propane [c]	74 g CO_2-eq MJ^{-1}	(JRC 2011)
Natural gas [a]	69 g CO_2-eq MJ^{-1}	(JRC 2011)
Fertilizer production – N	6.6 g CO_2-eq g^{-1}	(Börjesson et al. 2010)
Fertilizer production – P	2.9 g CO_2-eq g^{-1}	(Börjesson et al. 2010)
Fertilizer production – K	0.4 g CO_2-eq g^{-1}	(Börjesson et al. 2010)

[a] Used in base case calculations.
[b] Used in sensitivity analysis for electricity or heat demand in biogas production.
[c] Propane required for spiking the biogas replaces natural gas in the grid.

Table 4. Product GHG emissions used in systems expansion

Emissions from replacing crops or fossil fuels	
Oats grain	407 kg CO_2-eq (t DM)$^{-1}$
Rape seed	829 kg CO_2-eq (t DM)$^{-1}$
Wheat grain	407 kg CO_2-eq (t DM)$^{-1}$
Fossil fuels	84 kg CO_2-eq MJ^{-1}

Direct biogenic emissions of N_2O were calculated based on the IPCC model, where 1% of the N contained in crop residues is assumed to be converted to N_2O-N (IPCC 2006). This IPCC N_2O emission factor (EF) of 1% was also applied for the N added through biofertilizer, while a national EF of 0.8% (Naturvårdsverket 2013) was used for mineral fertilizer in the base case, and the IPCC EF in the sensitivity analysis.

Nitrogen losses to air after field application were calculated as corresponding to 15% of digestate content of NH_4-N when applied in cereals/rape and 30% when applied in grass-clover (Naturvårdsverket 2013). The corresponding loss at mineral fertilizer application was 0.9% (Naturvårdsverket 2013). Nitrogen leakage to water was calculated based on regional data to 42, 43 and 35 kg N ha^{-1} a^{-1} for the crop rotations in the reference, GC1 and GC2 scenarios respectively (Johnsson et al. 2008). For indirect N_2O emissions, 1% of nitrogen emitted to air or water was assumed to be converted to N_2O-N (IPCC 2006).

The increases in SOC for all scenarios as presented in Part I (Prade et al. 2014) are in the present study given as average annual CO_2-uptake. In addition, N in a mass ratio of 1:10 to accumulated SOC is assumed to be made unavailable for biogenic N_2O formation in the soil due to the integration in soil organic matter.

Sensitivity analyses were performed for a range of the input assumptions as summarized in Table 5. For details on these assessments and the selection of data, please refer to Björnsson et al. (2013).

Table 5. Background data and assumptions in base case and sensitivity analyses

Variable	Base case	Sensitivity analysis
Time span	40 years [a]	20 years [b]
Amounts of crop residues	Nordic data [a]	IPCC method [b]
N_2O EF mineral fertilizer	0.8%, National [c]	1%, IPCC [b]
Digestate storage methane leakage	National MCF 3.5% [c]	Experimental for digestate [d]
Biogas production – electricity [e]	Swedish mix	Nordic mix
Biogas production – heat [e]	Natural gas	Wood chips

[a] (Björnsson et al. 2013)
[b] (IPCC 2006)
[c] (Naturvårdsverket 2013)
[d] (Rodhe et al. 2013)
[e] See Table 3

5. Results and discussion

The GHG emissions are shown in Table 6 together with the product outputs of crops and biogas given as average per ha for the crop rotations in the three analysed scenarios.

Table 6. GHG emissions per category and product outputs for the three analysed scenarios

SCENARIO:	Reference	GC1	GC2
GHG EMISSIONS [kg CO_2-eq ha^{-1} a^{-1}]			
CULTIVATION			
Fertilizer	1 393	1 103	980
Diesel	208	252	263
Materials	192	197	176
Machinery	92	107	94
BIOGENIC N_2O			
Mineral fertilizer	729	577	509
Biofertilizer	-	252	405
Crop residues	179	298	320
Indirect	155	179	170
Soil organic matter	-98	-271	-307
BIOGAS PRODUCTION			
Process and pre-treatment energy	-	55	99
Upgrading energy and propane	-	30	54
Process methane leakage	-	25	45
Upgrading methane leakage	-	87	155
Digestate storage	-	34	79
SOIL ORGANIC CARBON (SOC)	-769	-2 124	-2 401
NET EMISSION	**2 081**	**801**	**643**
PRODUCT OUTPUTS			
Oats [t DM ha^{-1} a^{-1}]	1.00	0.80	0.67
Winter oil seed rape [t DM ha^{-1} a^{-1}]	0.63	0.50	0.42
Winter wheat [t DM ha^{-1} a^{-1}]	3.25	2.60	2.17
Biogas [GJ ha^{-1} a^{-1}]	-	17.3	30.9

The avoided GHG emission due to carbon sequestration by SOC incorporation has a strong impact on total GHG emissions in all scenarios, but especially in the GC scenarios, where grass-clover root biomass and biofertilizer give additional SOC contributions. In the reference scenario, the crop residues left in the field give a low and slow build-up of SOC content to 2.9% after 145 years, while in the GC scenarios, 3% SOC is reached already after 30 years, with a steady state level after 125 years of 4.1-4.4% (Prade et al. 2014). Emissions related to inputs in cultivation (diesel, fertilizers, materials, machinery) decrease in the GC scenarios due to the decreased fertilizer demand when biofertilizer is used. Biogenic N_2O emissions, however, increase due to larger emissions from biofertilizer and from the nitrogen in crop residues from grass-clover. At the same time, the SOC build-up in the GC scenarios also gives a large incorporation of N into soil organic matter, which decreases the N_2O emissions, giving a similar net emission of biogenic N_2O in all scenarios. Emissions related to the production of biogas in the GC scenarios give a relatively small contribution. All in all, the conventional food crop rotation gives an average GHG emission of 2.1 t CO_2-eq ha^{-1} a^{-1}. The emissions from the GC scenarios with integrated food crop and grass-clover production are much lower, 0.6 to 0.8 t CO_2-eq ha^{-1} a^{-1}.

In the systems expansion, the 20-33% lower production of food crops in the GC scenarios (Table 6) is included as an added climate impact from cultivation of these crops elsewhere in the region, while the avoided GHG emission when the upgraded biogas (Table 6) replaces fossil vehicle fuels is subtracted. The resulting total GHG emissions after systems expansion are shown in Figure 1, where the striped bars show the emissions summarized per category from Table 6, and the dotted bars show the impact of systems expansion. The resulting net GHG emission is negative for both GC scenarios; -0.2 t CO_2-eq ha^{-1} a^{-1} when one year (GC1), and -1.2 t CO_2-eq ha^{-1} a^{-1} when two years of grass-clover crops is integrated in the food crop rotation (GC2).

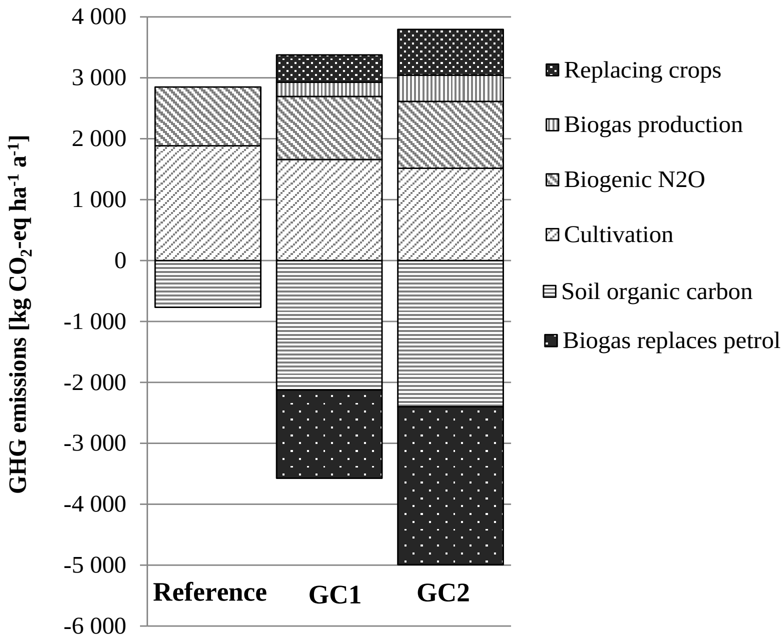

Figure 1. GHG emissions summarized per category in Table 6 (striped bars), and after systems expansion (dotted bars).

The net GHG emissions per scenario from Figure 1 are shown as the base case in Figure 2, together with the impact of changing input parameters in SOC modelling, cultivation and biogas production in the sensitivity analysis. In the base case, the annual SOC change was calculated over a time span of 40 years, to reflect the slow process (125-145 years) of achieving steady state SOC levels. IPCC suggest a 20 year timespan, which is applied in the sensitivity analysis and give lower annual GHG emissions for all scenarios. In the base case calculations, amounts of crop residues are calculated based on Nordic data on ratios between harvested crop and crop residues (Björnsson et al. 2013; Prade et al. 2014). The IPCC methodology evaluated in the sensitivity analysis differs mainly in suggesting much higher cereal straw yields (IPCC 2006). IPCC data thus give higher SOC accumulation and lower GHG emissions for all the investigated scenarios, but with larger impact on the reference scenario with 75% cereals in the crop rotation. The straw yields suggested in the IPCC calculation model are, however, unrealistically high compared to actual straw yields in cereal cultivation in Sweden (Nilsson and Bernesson 2009). The only other aspect in the sensitivity analysis with a noticeable impact on the net result is the assumption about increased methane emissions from digestate storage, which is based on an experimental study on digested cattle manure (Digestate storage CH_4, Figure 2). The risk of methane leakage is an important aspect to consider, since digestate from grass-clover digestion will contain quite a high residual amount of undigested organic material. This makes the digestate interesting as biofertilizer, but also increases the risk of methane leakage during storage. Since 99% of the digestate methane leakage has been shown to occur during the warmer months of the year, (Rodhe et al. 2013), a way of minimizing leakage is to make sure the storage is emptied in spring, which is what is assumed in the base case in the present study.

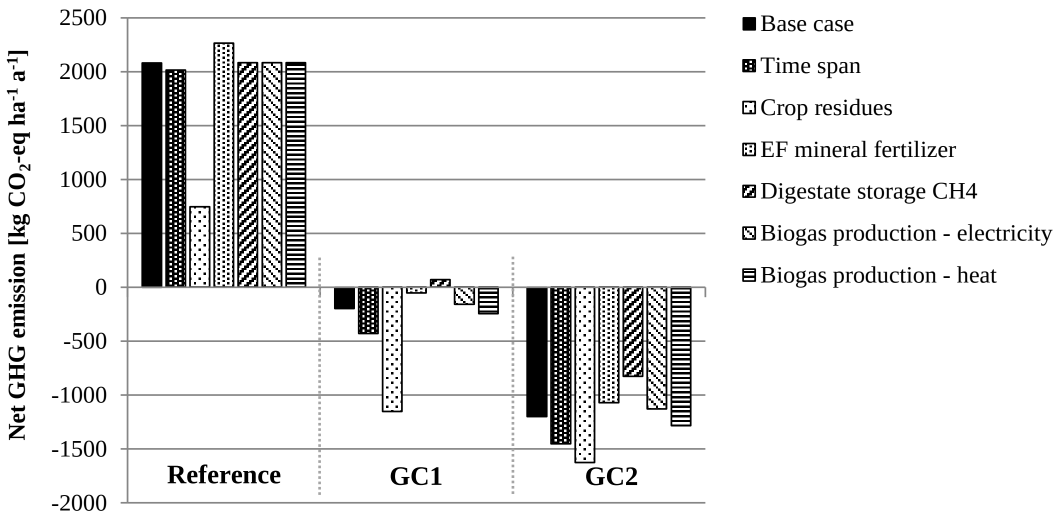

Figure 2. Net GHG emissions for the three analysed scenarios in the base case (black bars) and after varying a range of input parameters in the sensitivity analysis.

Looking at the difference between emissions in the reference scenario and the investigated GC scenarios, the shift from the 4-year cereal dominated food crop rotation to a 5-year crop rotation that includes one year of grass-clover as an energy crop for biogas production would result in avoided emissions of 2.3 t CO_2-eq ha^{-1} a^{-1} (ranging from 1.9 to 2.4 t CO_2-eq ha^{-1} a^{-1} based on the variation shown in the sensitivity analysis, Figure 2). Shifting to a 6-year crop-rotation with two years of grass-clover would give avoided emissions of 3.3 t CO_2-eq ha^{-1} a^{-1} (ranging from 2.4 to 3.5 t CO_2-eq ha^{-1} a^{-1}, Figure 2).

6. Conclusions

Combining food production with renewable energy production has been suggested as one possible approach to achieve food systems with lower GHG emissions and to combine food security with energy security (FAO, 2011). The studied change from a cereal dominated food crop rotation to a system with integrated production of food and energy crops was shown to strongly reduce GHG emissions from the cultivation system. Since production of energy crops on farm land is sometimes seen as conflicting with sustainable food crop production, it is important to also highlight cases where an integrated approach can have multiple benefits. The possibility of using grass-clover crops for biogas production opens up for a possibility of integrating grass-clover in the crop rotation in regions with no demand for cattle feed. Grass-clover crops will diversify the crop rotation and have a strong impact on the build-up of SOC. Such a change in cultivation practice could in turn improve long term soil fertility at the same time as giving significantly decreased GHG emissions, in the range of 2-3 t CO_2-eq ha^{-1} a^{-1}, for the investigated cultivation system.

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Building consensus on a generic water scarcity indicator for LCA-based water footprint: preliminary results from WULCA

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ABSTRACT

Consuming water can affect human health (e.g. by reducing availability of irrigation water and hence food availability), ecosystems (by decreasing water availability for terrestrial/aquatic species) and future generations (by depleting non-renewable resources). However, no standard method exists to quantify the stress on water without favoring any of these areas of protection. Stress/scarcity indexes have focused on an anthropocentric perspective, and a few on an ecocentric perspective. We explore the possibility of developing an indicator considering the water resource as a whole and propose a method which is not centered on an area of protection but rather assesses the extent to which all water demand and availability differ within a watershed (i.e. hydrocentric). This concept can eventually serve as a single metric to assess potential impacts from water use and be used consistently in the application of the upcoming ISO standard and for eco-labelling of food and energy products.

Keywords: water use impact assessment, Water Footprint, environmental impacts, water stress, water scarcity

1. Introduction

Life Cycle Assessment (LCA) has served as a decision-making tool to help reduce environmental impacts for several decades. Recently, the methodology was used to assess water-specific impacts and group them in a new water footprint concept, currently being framed in an upcoming ISO standard (ISO/FDIS 14046, 2014). While this document provides principles, requirements and guidelines, no specific characterization method is recommended even though several have emerged in the last five years to assess impacts from water use (Kounina et al. 2013). The need for a consensus-based method is clear and particularly relevant for food production systems where product-level environmental labels and declarations are already emerging. In 2013, the UNEP-SETAC Life Cycle Initiative recognized the need for a consensual method following the many existing water use impact assessment methods described by Kounina et al. (2013), and solicited the Water Use in LCA Working Group (WULCA), fostering method development and applications since 2007, to undertake this task.

By bringing together method developers and experts from the fields of LCA, hydrology, ecology, etc., WULCA will propose by the end of 2015, consensus-based indicators to assess impacts from water use, which can be used to comply with the requirements of the ISO document. This paper presents the water footprint concept as defined in the upcoming ISO 14046 standard, and WULCA's progress towards a consensus-based method. As agriculture accounts for over 70% of water withdrawal and its water consumption is estimated to increase by a 20% by 2050 (WWAP, 2014), combined with the fact that agri-food global businesses are expanding and hence putting new stress on local aquifers (WBCSD, 2012), the food sector is benefitting most from such methodological developments.

1.1. LCA and water footprinting

In contrast with an LCA, an LCA-based water footprint is the fraction of LCIA results which are related to water resource. These LCIA results include impacts associated with water use, and the subsequent effect on water availability for humans and ecosystems, as well as direct impacts on the water resource and its users from relevant emissions to air, soil and water (see Fig.1). These latter impacts are quantified using traditional LCA impact categories (e.g. freshwater eutrophication, freshwater acidification, human toxicity and eco-toxicity).

A water footprint may be presented as the result of a stand-alone assessment or as a sub-set of results of a larger environmental assessment, such as an LCA. As per the ISO standard, a qualifier is used when a water footprint study is limited to certain aspects only. A “water scarcity footprint” and a “water availability footprint” assess impacts associated with water use only, whereas a water footprint (no “qualifier”) assesses all relevant impacts related to water, hence including relevant emissions that occur without any water use and yet still impact water (e.g. SO_x emissions to air causing freshwater acidification).

In the recent years, businesses have started to include scarcity indicators (sometimes also called stress indicators) to assess potential impacts of water use. In the context of the standard, this is the equivalent of a “water scarcity footprint”, or a “water availability footprint” (if the change in water availability caused by water degradation is also included). However, at this point no consensus-based approach exists for doing so and results are not always comparable when different scarcity/stress indicators are used for characterization.

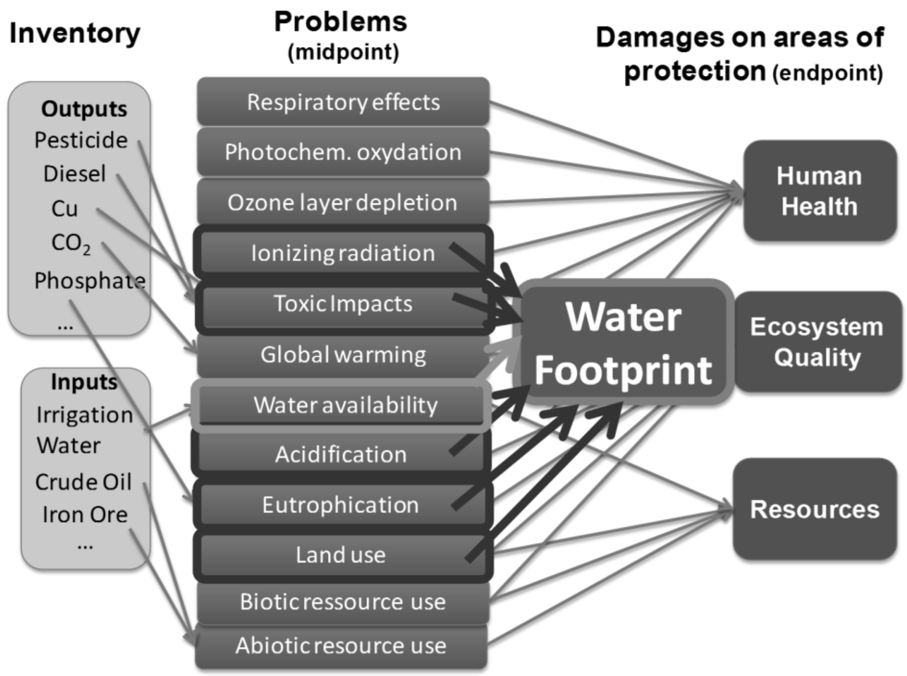

Figure 1: Representation of a water footprint with respects to LCA impact categories (figure from Impact World + (Bulle et al. 2014)).

1.2. Scarcity

From the ISO definition, scarcity is the “extent to which demand for water compares to the replenishment of water in an area, such as a drainage basin” (ISO 14046 2014). Up until now, water scarcity indexes have been built to reflect the problem as one for either humans or ecosystems. Most indexes (Pfister et al. 2009; Boulay et al. 2011; Frischknecht et al. 2009; Gassert et al. 2013) considered only human water use divided by water availability (see Eq.1). A few others (Smakhtin et al. 2004; Hoekstra et al. 2012) have considered that water scarcity should exclude water requirement for ecosystems to be maintained in “fair” condition, resulting in an index that shows whether the human’ water use impairs ecosystems (>1) or not (<1) (see Eq.2). Hence, none of these indicators represents a neutral assessment of the relationship between all water demand and availability.

The main gap is that no indicator exists where demand for water in an area is compared with availability in such a way that modeling choices and interpretation of the physical meaning are explicit and transparent.

$$Scarcity_{Anthropocentric} = Fn\left(\frac{human\ water\ use}{water\ availability}\right) \quad \text{Eq. 1}$$

$$Scarcity_{Ecocentric} = Fn\left(\frac{human\ water\ use}{water\ availability - ecosystem\ water\ requirement}\right) \quad \text{Eq. 2}$$

1.3. Consensus-based indicator project from WULCA

WULCA commenced a new two-year activity in January 2014 aiming at developing consensus-based indicator(s) for water use impact assessment. The first steps defined a framework (based on previous WULCA deliverable (Bayart et al. 2010; Kounina et al. 2013)) and led to the identification of the three sets of indicators on which to focus (see Fig.2): 1) The impact pathway leading to damages on human health is already modeled by different methods and ready for harmonization (Boulay et al. 2014). Hence it was included in the work and a consensus-based methodology defining this impact pathway is under development. 2) The ecosystem impact pathway includes several methodologies with possible complementary assessments (Kounina et al. 2013). A sub-group was formed to harmonize the different impact pathways in this category, by defining a consistent framework and possibly identify a midpoint indicator early in the impact pathway. 3) Answering the demand from industry for a generic (not human- or ecosystem-oriented) and recommended stress/scarcity indicator, a specific sub-group focused its efforts on the development of such a metric. It was set out as an indicator independent of other impact pathways and not leading directly to any endpoint damages. Further harmonization with the conventional midpoint-endpoint framework may come later as the indicators further develop. This paper is presented by the sub-working group focusing on this last stress-based midpoint indicator and presents results of the findings to date. The impact pathways leading to impacts on resources or compensation processes were considered not sufficiently mature to be included in this consensus building phase.

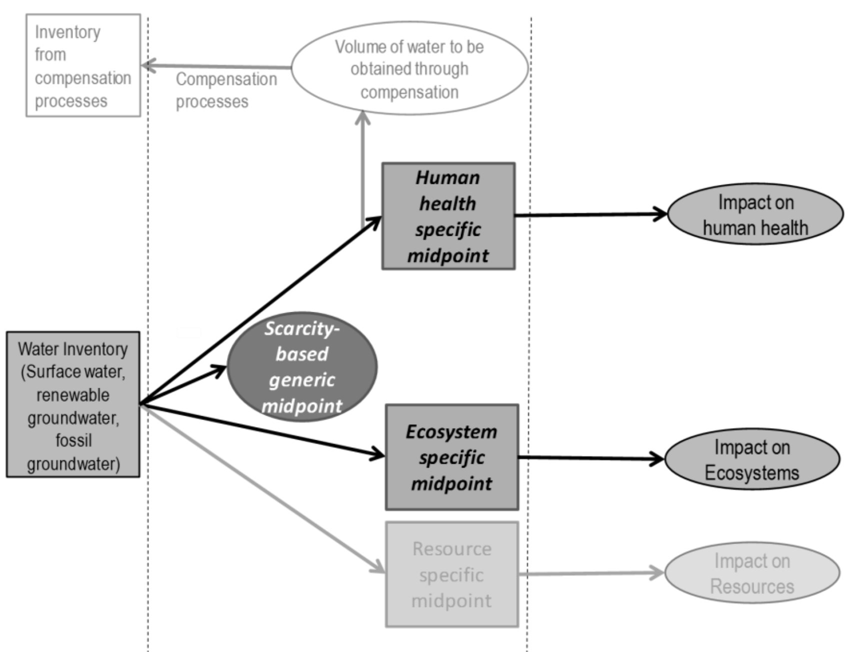

Figure 2. Chosen impact pathways for WULCA's consensus building work.

2. Methods

In order to provide a scarcity indicator which represents a generic assessment of the demand and availability of water in a region, we define scarcity following ISO/FDIS 14046 definition, as shown in Eq.3, where the total water demand includes all users, human and non-human, and water availability includes the available renewable resource. This indicator is therefore not anthropocentric or ecocentric, but simply based on the water resource, i.e. hydrocentric.

$$Scarcity_{hydrocentric} = Fn\left(\frac{Total\ water\ demand}{Renewable\ water\ availability}\right) \quad \text{Eq.3}$$

To illustrate the difference in scarcity definitions, we compare the three approaches using eq. 1-3 without a scaling function (just the ratios) in a simple preliminary assessment. Not all necessary high quality data was available at this early stage of the project and proxy data was used to illustrate the concepts. Human water consumption (HWC) from WaterGap (Alcamo et al. 2003) was used to represent water demand from humans. Environmental water requirements (EWR) assessment from Smakthin et al. (2004) were used to represent ecosystem water demand. These values represent the portion of the available flow required to maintain ecosystems in a "fair" condition, defined by the authors as "moderate or considerable modification from the natural state where the sensitive biota is reduced in numbers and extent". This state represents a modeling choice which should be agreed on and assessed with corresponding data. This data is regionalized and readily available for the entire globe, and was used as a first assessment to illustrate the concepts; it does not yet include terrestrial ecosystems water requirements and may not represent the best data available. Water availability (WA) data from WaterGap (Alcamo et al. 2003) was used for this first assessment.

3. Results

Figures 3, 4, and 5 illustrate the different scarcity ratios: anthropocentric, ecocentric and hydrocentric, respectively. They are calculated using the ratios presented in Eq. 1, 2 and 3 respectively. The difference between anthropocentric and ecocentric scarcity is negligible for most of the world, with few watersheds showing visibly higher scarcity in the ecocentric case (hardly visible on a black and white scale). This makes sense, since the scarcity level is increased by environmental flow requirements but the pattern is not changed.

Not surprisingly, results on Fig.5 show higher scarcity in most regions of the world for the hydrocentric indicator since this includes ecosystem water demand within the numerator of the equation and therefore also regions with no human water consumption show some scarcity. Figure 6 shows the difference in results between the anthropocentric indicator and hydrocentric indicator. At a minimum, scarcity increases in comparison to the anthropocentric approach between 0.2-0.3 around the world. Most of North and South America, Europe and Central Asia increases between 0.3-0.4, and some watersheds in the USA and Eastern Europe increase by up to 0.47. This difference represents exactly the EWR percent of available water assessed by Smakhtin et al. (2004), which varies from 0-47%. By integrating ecosystem's water requirement, this approach therefore gets rid of the artifact that water scarcity in desert areas is 0 if no human is using water, as discussed in Berger et al (2014).

Figure 7 shows the difference in results between the hydrocentric approach and the ecocentric indicator. In this case, some highly scarce regions show a higher indicator with the ecocentric indicator than with the hydrocentric approach, but these regions are still consistently at the top of the scarcity scale. As a mathematical demonstration, if one hypothetical region shows a water availability of 100 m^3, with 90% of this amount being consumed by humans (90 m^3), yet having a EWR value of 40% (40 m^3), the ecocentric indicator would result in 90/(100-40) = 1.5, indicating that the water use is tapping into the ecosystem's requirements. A hydrocentric approach results in (0.9+0.4)/100 = 1.3, indicating the shared pressure on both human and ecosystem needs for water.

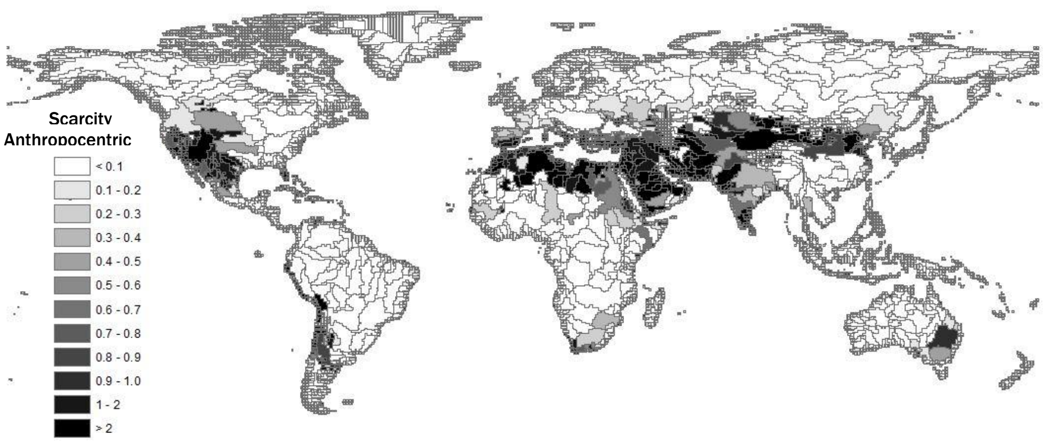

Figure 3. Anthropocentric scarcity assessment showing the ratio of consumed water by humans to available water (HWC/WA).

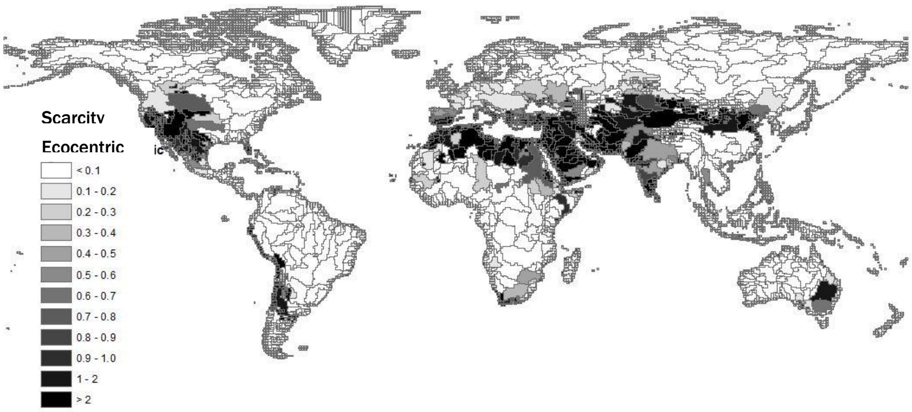

Figure 4. Ecocentric scarcity assessment showing the ratio of consumed water by humans to available water, to which water for ecosystem has been subtracted (HWC/(WA-EWR)).

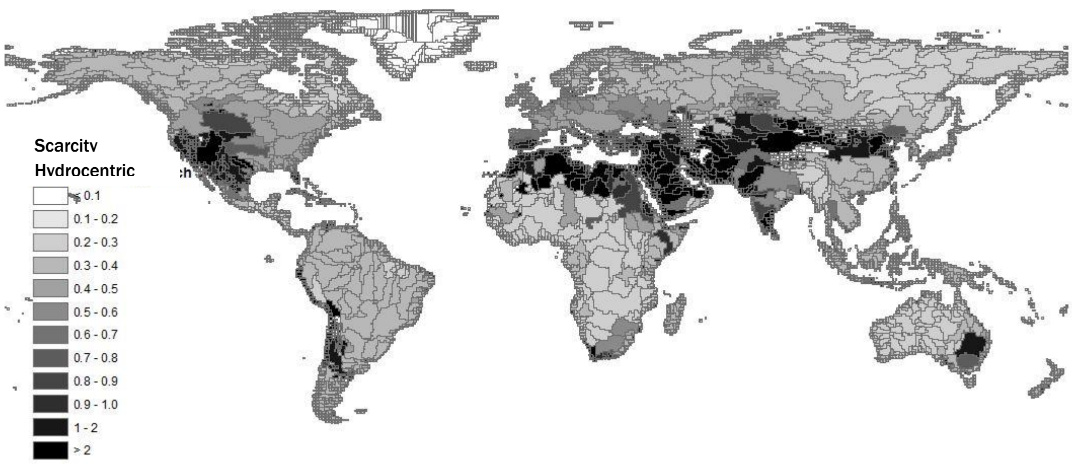

Figure 5. Hydrocentric scarcity assessment based on the ratio of all water demand to all water available, showing the pressure on the water resource ((HWC+EWR)/WA).

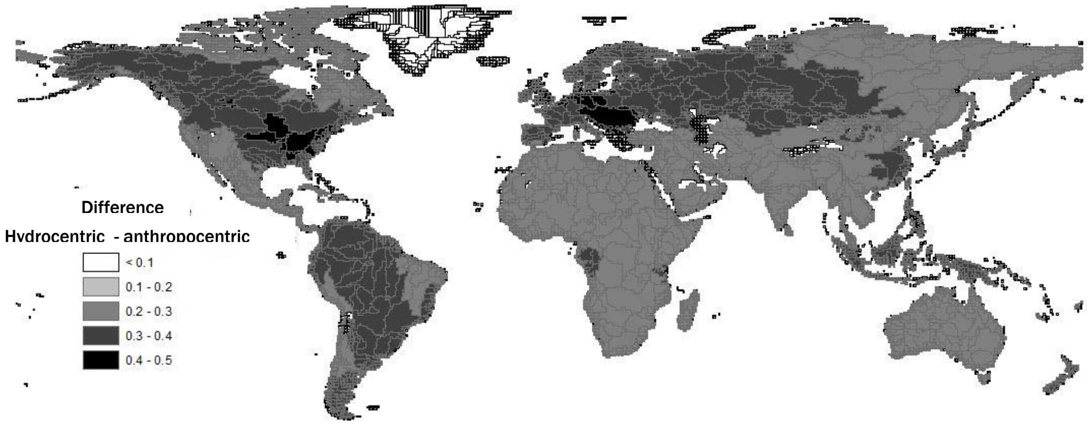

Figure 6. Difference (decimal percentage) between the hydrocentric approach as shown in Fig.5 and the anthropocentric scarcity shown in Fig.3.

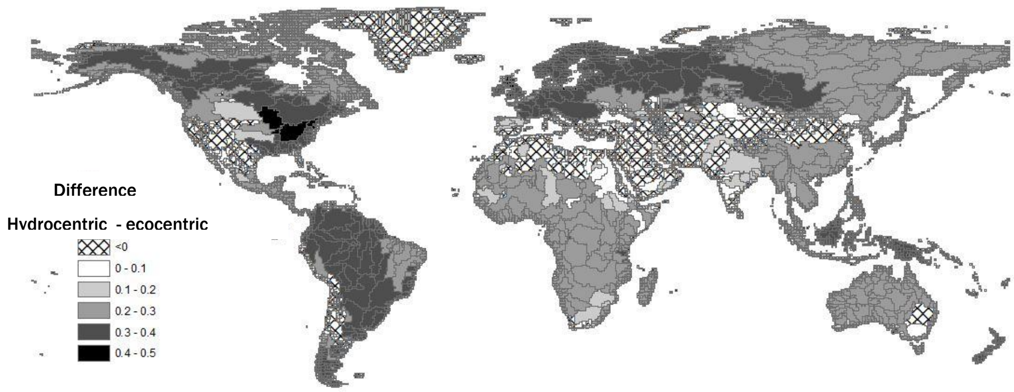

Figure 7. Difference (decimal percentage) between the hydrocentric approach as shown in Fig.5 and the ecocentric scarcity shown in Fig.4.

4. Discussion

Results presented in this paper show intermediate work as the working group progresses towards a consensual indicator. At this point, we have identified that scarcity indicators used until now have been focused mainly on potential impacts on humans, and some towards ecosystems, but none reflected simply the pressure on the resource coming from the demand of all local water users in comparison with the availability. At this point, potential impacts on ecosystems from water use have only been accounted for at the endpoint level (e.g. Pfister et al. 2009, Hanafiah et al. 2011). A combination of human and ecosystem impacts based on endpoint methods and subsequent weighting of these indicators have been suggested by Ridoutt and Pfister (2013). However, such aggregated endpoint-based CF integrate a weighting scheme, which is only allowed by ISO as an additional and subsequent step following impact characterization, after first obtaining separated impact category indicator results. This ensures that value choices are transparently provided and eventually can be adapted to the values of the decision maker. While such weighted methods are still being investigated (Ridoutt and Pfister, 2014), the objective of the approach presented in this paper is rather to provide a generic indicator representing potential problems associated with water use in relation with the local scarcity, without focusing on one area of protection.

Findings of this paper indicate that an indicator that assesses scarcity following the ISO definition and considering all water demand can provide an alternative perspective and better reflect the potential consequences of using water in different areas around the world. Hydrocentric scarcity values are higher than anthropocentric scarcity values and generally higher than ecocentric scarcity values (except in highly (eco)scarce regions), and hence may allow the quantification of potential problems in such regions that have been excluded so far on the scarcity level.

However, the main challenges lie in the proper attribution of ecosystem water needs compared to human water consumption and the data availability to develop a global indicator of reasonable quality for robust assessments. In this paper, human water consumption was used, along environmental water requirements for ecosystems based on “fair conditions”, which is not based on the same reference and hence still introduces some bias. Results would likely differ and scarcity values increase if the same perspective on “demand” is taken. However, this was performed as a preliminary assessment with readily available data, and defining and assessing the water demand from humans and ecosystems with the same underlying basis (e.g. actual water demand, ideal/pristine water demand, etc.) is necessary to reduce bias and remain transparent. Best available data should be used to represent the water demand both from a human and an ecosystem perspective, including aquatic and terrestrial ecosystems, with special attention to wetlands (Verones, Pfister, et al. 2013; Verones, Saner, et al. 2013). Different definitions and sources of data exist to quantify water availability, whether current or pristine for example, and this aspect should be investigated and the most relevant source for this indicator chosen. Also, while this work has shown only blue water-related data, interactions with green water and the change in blue water associated with land use change should be investigated. Coherence among the sources of data, the geographic and temporal resolution, and close analysis to prevent double counting are important aspects that should be analyzed.

5. Conclusion

This work provided an overview of the relationship between LCA, water scarcity, water availability and water footprinting, in light of the upcoming ISO standard 14046. It quantifies the characterization factors for different interpretations of the scarcity concept and applied directly the ISO definition to provide a first glance at a generic hydrocentric indicator of water scarcity quantifying the pressure on the water resource in a region by relating the water demand and availability. Results show higher scarcity in comparison with the currently used anthropocentric methods as it encompasses a larger problematic. The WULCA working group is further developing this indicator through ongoing investigating of data sources, hypothesis, data resolutions and choices associated with this novel approach.

6. Acknowledgements

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From wheat to beet – challenges and potential solutions of modeling crop rotation systems in LCA

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ABSTRACT

One of the remaining methodological challenges of agricultural LCAs is the inclusion of crop rotation effects. They are caused by changes in physical, chemical and biological properties of the agricultural land over time (e.g. soil structure, presence and availability of nutrients, phytosanitary conditions) and cannot be easily measured. LCA studies, focusing on only one vegetation period, have a limited ability to include those crop rotation effects, unless explicit modeling measures have been taken to include them. A summary on existing approaches is given and remaining gaps are identified. A new approach for the modeling of crop rotation effects is suggested. It includes the alignment of system boundaries with the whole crop rotation system, a systematic description of crop rotations, the inclusion of all inputs and all outputs and as a last step, allocation to the crop specific burdens using the agriculture-specific Cereal Unit.

Keywords: Agriculture, LCA, Crop rotation, System boundary, Cereal Unit allocation

1. Introduction

One particular challenge for proper description of agricultural reality is the consideration of crop rotations in LCA. Crop rotation describes the sequence of different agricultural crops, grown on the same field. “If an LCA study focuses on just one crop (such as wheat) it fails to account for the interactions between this crop and preceding and subsequent crops.” (Cowell et al., 1995)

Features of crop rotations are explained, several examples for positive crop rotation effects are mentioned and the need for including crop rotation effects in LCA is derived from this. Different approaches for including those effects are described, but none of them is completely satisfactory. A consensus on a uniform approach has not been achieved yet – even though crop rotation effects are physical reality, are described in agricultural publications and do have strong influence to the agricultural practice, e.g. cultivation planning, plant protection and plant nutrition.

1.1. Features and the need for inclusion of crop rotations in LCA

By means of growing different crops in chronological sequence, positive effects from the current to the succeeding crop can be achieved, e.g. improvement of phytosanitary conditions (hereby avoiding diseases) or the improvement of nutrient availability of succeeding crop (by using different nutrients or leaving different nutrients in residues or sourcing the nutrients from different soil horizons). These crop rotation effects can be physically measured by long term field experiments, are well described in agricultural publications and are very important for the agricultural practice, e.g. in terms of crop planning and supply of nutrients to plants (Blanco-Canqui and Lal, 2009; Forsyth, 1804; Russell and Russell, 1973; Wirghtson, 1921). “Use of crop rotations can help maximise productivity because different crops and farming practices have varying impacts on the soil, affecting properties such as fertility, texture, structure, population of microorganisms, number of weed seeds and so on.” (Cowell et al., 1995) Zegada-Lizarazu and Monti summarized several advantages of crop rotations:

- “Enhanced soil fertility and higher yields
- Improved soil structure and maintenance of long-term productivity and organic matter.
- Longer period of land cover with subsequent lower erosion.
- Reduced use of agricultural inputs such as agrochemicals and synthetic fertilizers [lower disease pressure due to changing crops].
- Diversified production with greater marked opportunities and lower economic and climatic risks.
- Increased biodiversity and less monotony of the landscape.
- Time-diluted farming activities [seedbed preparation, sowing process, plant protection, harvesting].” (2011)

1.2. Existing approaches to include crop rotation effects in LCA and their limitations

There are several approaches to consider the shift of nutrients from a current crop to the subsequent crop in agricultural LCAs. But in detail, the existing approaches differ. For the nutrients phosphorous (P) and potassium (K), corrections are made for residues that either remain on the field – by deducting the environmental impacts from the total impacts and allocating it to the subsequent crop (Nemecek et al., 2011; Nemecek and Schnetzer, 2011) – or removed from the field – by allocating the burden to those harvested co-products (Nemecek and Kägi, 2007; Nemecek et al., 2008). For the nutrient nitrogen (N), remaining in crop residues on the field, a credit can be given if reduced a fertilizer dose is recommended for the subsequent crop (Nemecek et al., 2011). Van Zeijts et al. suggest to allocate N completely to one crop; to allocate P and K according to the uptake and uptake efficiencies of the various crops; to allocate the application of organic matter according to their land use share in the cropping plan. Furthermore, they recommend to intensively study each agricultural activity in the cropping plan and to decide whether done for individual crops or more than one crop (1999). Van Zeijts et al. underline, that agricultural activities, e.g. fertilization or crop protection, are typically "meant to benefit more than one crop" (1999).

"If an LCA focuses on just one crop (such as wheat) it fails to account for the interactions between this crop and preceding and subsequent crops." (Cowell et al., 1995) If agricultural fertilization is studied using a limited observation period – e.g. one single vegetation period – there is a certain probability for overlooking several aspects of the overall fertilization strategy for the studied agricultural farm (Flisch et al., 2009). "This raises the question of whether it would be more appropriate to draw a system boundary around a crop rotation rather than a particular crop." (Cowell et al., 1995) Cowell et al. argue, the farmed soil should be within the system boundary, "because it is an integral part of the production system" and even the soil does not cross the spatial system boundary, the soil quality must be taken into account (Cowell et al., 1995). Audsley et al. add, that the soil crosses the temporal system boundaries, and thus needs to be considered in LCA (Audsley et al., 1997).

The complexity of effects between elements in a crop rotation can be understood by studying GRUDAF (Grundlagen für die Düngung im Acker und Futterbau; Principles for fertilization in arable and fodder production). GRUDAF has been developed and released by the Swiss agricultural research institutes Agroscope Changins-Wädenswil ACW and Agroscope Reckenholz-Tänikon ART (Flisch et al., 2009). GRUDAF contains natural science based recommendations for the fertilization of arable crops and fodder. The document is oriented towards agricultural advisory service and farmers in order to assist the development of economically and ecologically sound fertilization strategies. In order to be in line with latest scientific knowledge and most recent production technologies, it is updated regularly (Flisch et al., 2009). The document reveals the complexity of fertilization and offers an excellent view into the vast number of dependencies of fertilization planning and concrete fertilizer amounts. Based on natural science based long-term experiments, Flisch et al. describe the following agricultural management aspects and soil properties that shall be considered for determining the fertilizer amount:

- Management aspects: Crop rotation design, Types previous crops, Usage of intermediate crop, Crop residue management, Number of grassland cuts or gazing of pasture, Long-term effects of organic fertilization (correction factors for second year after application), Animal-type specific nutrient composition of organic fertilizer, Consideration of organic farming practices, Amount of precipitation during several time periods (e.g. outside vegetation period).
- Soil properties: Mineralized nitrogen content, Soil organic matter, Humus content, Clay content, Soil skeleton, Nutrient content (nutrient supply categories A – E), pH value, Soil depth (shallow to deep), etc. (2009).

For each of those aspects, numerical correction factors are provided for adapting the actual fertilization practice (Flisch et al., 2009). Many of these aspects need to be considered on a broader time horizon, than just one vegetation period. This reveals that the nutrient availability and the uptake of individual crops are not only determined by fertilization activities, taking place after seedbed preparation. Instead, activities taking place months and even years before growing the considered crop, significantly affect the quantity and quality of the respective crop (Flisch et al., 2009). Because this situation applies for all different agricultural crops, one could state, the same error is acceptable for all agricultural crops – but one must acknowledge, each agricultural crop has individually different nutrition requirement profiles.

Besides the removal of crop residues from the field and hereby affected presence of nutrients, many further aspects contribute to crop rotation effects. For example the change of crops help to reduce phytosanitary stress. The use of nutrients from different soil horizons and the improvements in soil structure lead to improved soil fertility and higher yields. These positive effects are plant-specific and have been proven by long-term field experiments.

Even though previously described approaches are suitable for integrating the nutrient shift from one to the succeeding crop into LCA, they fail to integrate the whole range of positive crop rotation effects. Improved phytosanitary conditions, reduced need for agrochemicals and increased yields are not covered at all. Because of their relevance to agricultural practice, we suggest including these effects in the LCA-methodology and propose a respective approach in the next sections.

2. Material and methods

This section contains descriptions of helpful methods for the application of the new approach for including crop rotation effects in agricultural LCAs.

2.1. Systematic description of crop rotations

In order to assess agricultural production systems, containing crop rotations or to carry out calculations with crop rotation elements, a systematic description of the crop rotation might be helpful. Such systematic mathematical representation and classification of crop rotations was carried out by (Castellazzi et al., 2008). General types of crop rotations are fixed rotation, flexible cyclical rotation with fixed rotation length, flexible cyclical rotation with variable rotation length and flexible non-cyclical rotation with variable rotation length; shown in Figure 1.

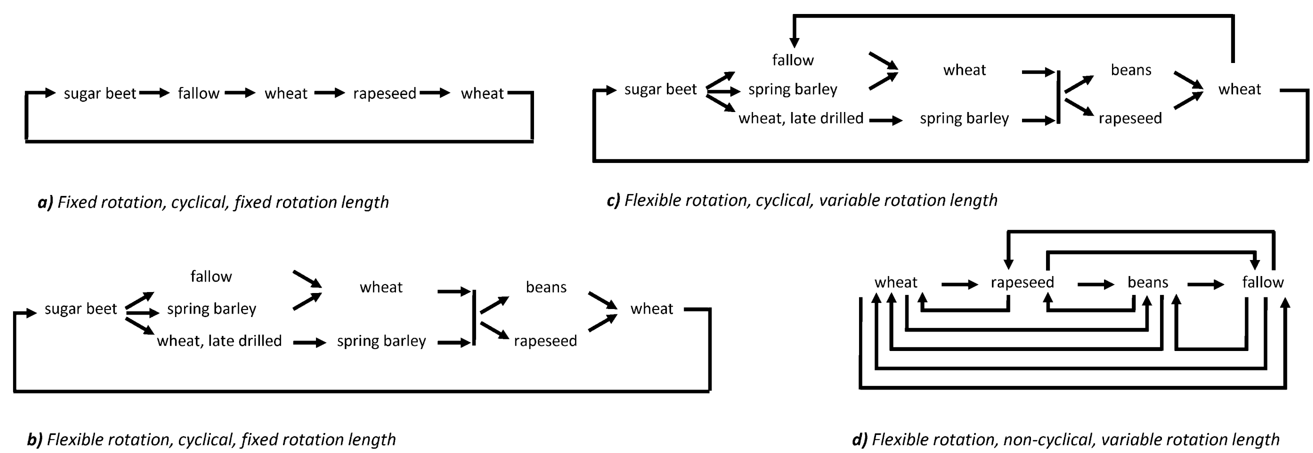

Figure 1. Examples of classified crop rotations; obtained from (Castellazzi et al., 2008)

2.2. By-product allocation approaches and system expansion

Besides a proper systematic description of crop rotations, an appropriate way of allocating environmental burden between various agricultural products and by-products is helpful. Additionally to existing allocation approaches, such as mass allocation (based on mass), energy allocation (based on lower heating value) and economic allocation (based on market prices), a biophysical allocation approach based on the Cereal Unit was proposed by Brankatschk and Finkbeiner (2014). The Cereal Unit allocation is based on the Cereal Unit; it "has been used as common denominator in agricultural statistics for decades ... [and] is valid for vegetable and animal products" (Brankatschk and Finkbeiner, 2014). Cereal Unit conversion factors are used to make all agricultural products and co-products comparable. These factors are calculated mainly based on the nutritional value for animals, because 80% of the agricultural area in the world is used to feed animals (Brankatschk and Finkbeiner, 2014; FAO, 2009).

Well known in the LCA community and recommended by ISO to avoid potential allocation steps, is the product system expansion approach. When applying system expansion, the product system is expanded "to include the additional functions related to the co-products" (ISO 14044, 2006).

In the next section it is described, how the advantages of changing system boundaries, the use of the Cereal Unit allocation and systematic representations of crop rotations can be combined in order to include whole crop rotations into one LCA and hereby automatically consider crop rotation effects.

3. Proposal for a new approach to include crop rotation effects in LCA

Within this section we propose an approach for the inclusion of the previously described crop rotation effects into the inventory analysis of environmental life cycle assessments (according to ISO 14040 and ISO 14044). The presented steps are not meant to be an exhaustive description for performing an LCA. Rather, they shall be understood as supplement to the existing steps in the inventory analysis of an LCA. We would like to emphasize that the below proposed modification of the system boundary is only relevant during the collection of data for the Life Cycle Inventory and not the LCA as such. The overall scope of the LCA, the functional unit and the reference flow of the LCA do not need to be changed.

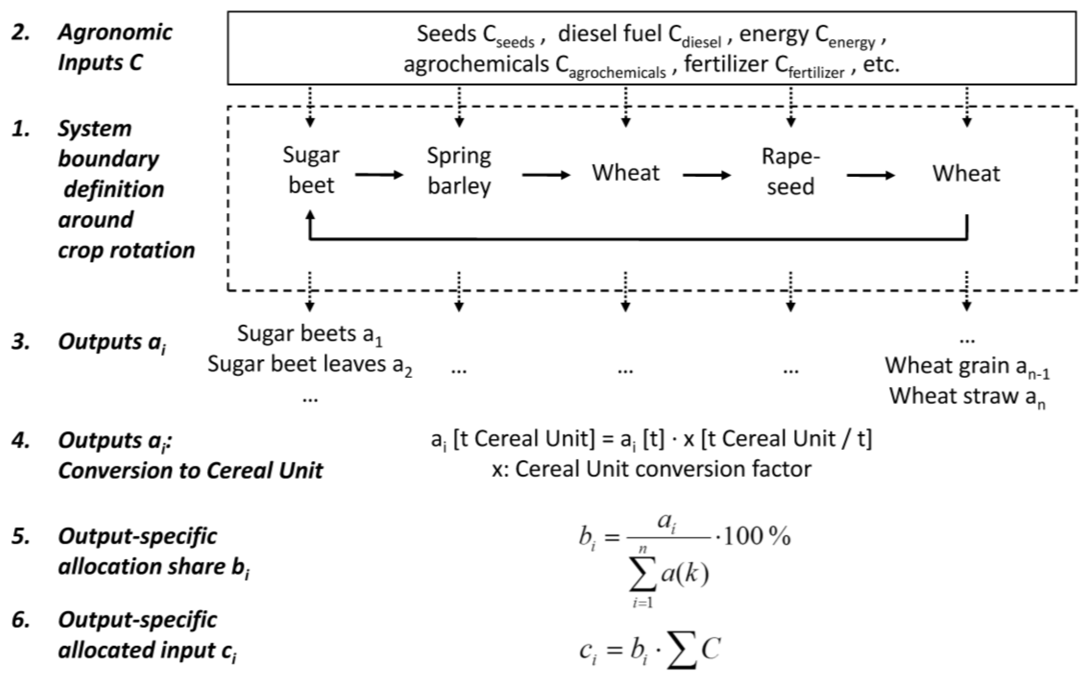

Figure 2. Calculation steps for inclusion of crop rotation effects in LCA inventory analysis

The proposed approach consists of the following steps (see also Figure 2): **First**, the assessed agricultural crop grows in a crop rotation. This crop rotation is identified and a concrete structure of this crop rotation is determined. The system boundary according to ISO 14040 is defined around this crop rotation. Hints for systematic and mathematical descriptions of crop rotations are given in the methods section and in the work from Castellazzi and colleagues (2008); examples are given in Figure 1. **Second**, all agronomic inputs (seed, diesel fuel, energy, agrochemicals, fertilizer, etc.) of the whole crop rotation cycle including all crop rotation elements are quantified. **Third**, all outputs (including products and by-products) of this crop rotation leaving the agricultural field are considered. **Fourth**, the units of all agricultural outputs are converted from metric tons into Cereal Unit tons. Necessary Cereal Unit conversion factors are available in several publications (BMELV, 2012; Brankatschk and Finkbeiner, 2014; Mönking et al., 2010). **Fifth**, allocation factors are calculated for all agricultural outputs of the whole crop rotation, using the amounts given in Cereal Units. Plausibility check: the sum of all allocation factors end up in unity or 100 %, respectively. **Sixth**, using the allocation shares, calculated in step five, the sum of each agricultural input (seed, diesel fuel, energy, agrochemicals, fertilizer, etc.) is allocated between all individual agricultural outputs.

As a result, the LCA practitioner obtains for each output a specific set of necessary inputs. Next steps of the LCA take place using these results. Further LCA steps are not affected by this approach.

4. Discussion

In current LCA practice, crop rotation effects are only partly included, because it is hardly feasible to quantify such crop rotation effects, e.g. it is enormously sophisticated to measure each nutrient flow in the soil – such data are not easily accessible and typically gathered in field experiments over decades. LCA today assesses each crop individually and not infrequently ignores these crop rotation effects, even though they are crucial for maintaining soil fertility and therefore are relevant for the sustainability of the whole system. Towards a solution for the consideration of these “hidden” nutrient flows and further hardly quantifiable crop sequence effects, we suggest a supplementation to the LCA methodology by adding an additional calculation step for the inclusion of the crop rotation effects into the Life Cycle Inventory according to ISO (2006). This supplemental approach is suitable for all agricultural LCAs and takes into account all inputs and all outputs of the crop rotation and thus includes as well inter-crop relations.

Within recent LCA-practice it is not obligatory to consider nutrient shifts from one to the subsequent crop. Hereby the fertilizing efforts are attributed to solely one crop of one vegetation period. This leads to free-rider situations for crops that consume nutrients, left by the previous crop on the field (e.g. by crop residues). In that sense, such crops do not get charged for their real nutrient-consumption, because they participate in the fertilization of the previous crop and subsequently other crops within the crop rotation carry more environmental load, than physically true. In other words, if LCA is performed for one individual crop, this crop automatically either enjoys the benefits or suffers the load of being part in a crop rotation. In this context it is worth to mention, that especially crops, leaving high amounts of nutrients in the soil might be systematically disadvantaged. This is the case, if the full amount of fertilizers, applied within their vegetation period, is accounted to them, even if they do not consume the full amount themselves and parts of their nutrients are transferred to the successor crop via their crop residues. By applying an adaption of the system boundaries to the crop rotation level, free-rider phenomena for nutrient-receiving crops and systematic disadvantages of nutrient-lending crops, can be avoided. Furthermore incentives are set for the inclusion of crops that perform nitrogen fixation into the crop rotation, because all crop rotation elements share the resulting environmental benefits.

The new approach for including crop rotation effects changes the system boundaries. Hereby all crop rotation elements and thus resulting crop rotation effects automatically are included in the LCA. This works, because nutrient flows between crop rotation elements do not cross anymore unconsidered any system boundary. Because the succeeding crop as well belongs to the same crop rotation and thus the nutrients remain inside the system boundary, their positive effect to the succeeding crop is considered. Those positive effects are not restricted to the immediately succeeding crop. As well improved conditions for further crop rotation elements throughout the whole crop rotation are included, due to the modified system boundary. This is especially relevant for phytosanitary effects that take effects over years and can be hardly directly measured. In case the complete crop rotation is assessed, less use of agrochemicals and improved yields are regarded.

The suggested alignment of the system boundary to the level of crop rotation brings LCA closer to the farmers’ perspective on crop rotation systems and his crop planning. The system boundary alignment leads to an immediate inclusion of all nutrient flows and changes in soil properties between crop rotation elements, without any need for measuring them. Furthermore the suggested method offers new capabilities to LCA to treat crops more fairly, especially for those crops that leave nutrients on the field for the succeeding crop.

The suggested approach also includes the Cereal Unit allocation approach as a relatively new allocation approach. It uses the Cereal Unit as an agriculture-specific biophysical unit that has been developed decades ago for the purpose of agricultural statistics and is being continuously updated and still used today in Germany. The Cereal Unit is mainly based on the feeding value of each agricultural product and via side calculation it is able to include even products, not directly fed to animals. By expressing all agricultural outputs of the inventory analysis in the same unit, they become comparable and computable. The Cereal Unit introduces by a common denominator to all agricultural products a comparability and computability for different agricultural outputs (Brankatschk and Finkbeiner, 2014). This allows allocating all agronomic inputs of the complete crop rotation to each individual agricultural output – independently if the output is vegetable or animal origin. In that sense it is a combination of system boundary alignment and biophysical allocation that allows a more realistic picture of agricultural coherences.

Even though changes in the system boundary are well known in LCA practice, they are not widely used in LCAs for agricultural systems. We see reason for that in the vast number of different outputs of one single system and enormous effort for handling so many diverging outputs. This is particularly the case when product-specific LCAs shall be done. The series of different agricultural products seems to be incomparable at the first glance. But the Cereal Unit as biophysical unit, that is suitable to depict all agricultural products and co-products, provides a powerful instrument to shoulder this task. The Cereal Unit allows sorting the number of different agricultural outputs – it makes different agricultural products and co-products comparable to each other and introduces computability. This serves as basis for an agricultural specific allocation approach. Within the Life Cycle Inventory, the Cereal Unit allocation approach is used to allocate all inputs uniformly to the individual outputs. Hereby, agricultural inputs that are applied in the vegetation period, but not used by the crop that was grown in the same time period to produce agricultural fruits, can be more fairly attributed to the overall crop rotation. The crop rotation elements that might be currently regarded as single players can be – by using this new approach – considered as team players and as well their interactions are taken into account.

5. Conclusion

A new methodology was proposed to supplement the established LCA methodology according to ISO 14040 series. A combination of aligning system boundaries in order to consider the whole crop rotation and the relationships between its elements and the Cereal Unit as basis for the allocation of the inputs to the individual outputs was considered to be appropriate. The new method allows to depict the agricultural system in appropriate time frames and includes fundamental agricultural coherences like crop rotation effects into the LCA methodology. Given examples were proven by agricultural coherences, that are scientifically described since hundreds of years and observed by humans since thousands of years, leading to recent agricultural production systems.

The presented method is a straightforward methodology to integrate whole crop rotations in agricultural LCAs, including crop sequence effects and establishing a performance-oriented allocation of environmental interventions to all agricultural outputs of the crop rotation. This new approach helps LCA models to draw a more realistic picture of agricultural reality and thus might lead to more credible results. The recommendations for agricultural systems derived thereof become more robust and help aiming the target of sustainable development.

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LCA of vegetarian burger packed in biobased polybutylene succinate

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ABSTRACT

Packaging preserves food quality and safety, but can determine a relevant part of the environmental impact associated with food consumption. A vegetarian burger has been selected as case study to evaluate, through LCA, the environmental performance of a food product packed with novel biobased polybutylene succinate (PBS), in comparison with traditional fossil based packaging (mainly polypropylene). The use of PBS instead of conventional plastics in primary packaging would still increase the environmental impact of the vegetarian burger. These results should, however, be considered in perspective: biobased PBS is in an early stage of development, and far from full industrial scale. Moreover, some first results from conservation and shelf life studies indicate that PBS packaging could extend the shelf life of some food products and thus presumably contribute to a reduction of food waste and associated environmental impacts.

Keywords: LCA, biobased PBS, bioplastics, shelf life, vegetarian burger

1. Introduction

Packaging plays a fundamental role in the food industry, allowing to preserve food quality and safety (Williams and Wikström, 2011). On the other hand, packaging production and disposal, in some cases, can determine a relevant share of the environmental impact associated with food consumption, and contributes to generation of waste and depletion of non renewable resources (Marsh and Bugusu, 2007). Many companies and organizations are currently developing and testing different kinds of bioplastics, in order to provide a more environmental friendly solution for packaging (Davis and Song, 2006). The SUCCIPACK project, promoted by the EU through the 7th framework programme, aims to support European industry efforts to introduce biobased polybutylene succinate (PBS) in the food packaging market (www.succipack.eu).

PBS is a semi-crystalline polyester bioplastic with 35-40% crystallinity, melting temperature of 114-115°C, glass transition temperature of -32°C, and similar properties to PET (Song et al., 2011). PBS is synthesized through polycondensation of succinic acid and 1,4-butanediol (BDO) (Rudnik, 2010) and its building blocks can be fossil based or biobased. Most of the PBS currently available on the market are fossil based or only partially biobased (Storz and Vorlop, 2013). Regarding its barrier properties, PBS is more similar to PLA than to polyolefins and can be considered as a middle oxygen barrier, and a middle/poor water barrier. Introducing PBS as a new biobased material in food packaging requires various innovations, such as optimization of its thermomechanical and gas barrier properties. In this context, the SUCCIPACK project aims at formulating entirely biobased PBS that can be flexibly applied in the food packaging sector, developing a bioplastic that is both biobased and biodegradable.

Using biomass resources as raw materials does not automatically guarantee a better environmental performance than fossil resources (Miller et al., 2007), due to, for example, environmental impacts of the agricultural phase or high energy consumption for processing, especially for new developed materials produced in small scale facilities. For this reason, life cycle assessment is applied to assess biobased PBS and PBS packaging solutions in a cradle to grave perspective, in order to avoid environmental trade-offs and contribute to the development of the bioplastic, providing ecodesign feedback through screening LCAs. LCA has allowed to identify the most critical aspects in the synthesis of biobased PBS, and to identify possible improvement options, some of which have been already implemented in the production process.

The focus of the SUCCIPACK project, along with the development of the material, is the introduction of new packaging solutions in the food packaging market; screening LCAs of existing food products have been conducted, in order to evaluate the environmental performance of new packaging solutions in comparison with existing traditional packaging. The study presented here reports the LCA results obtained for a vegetarian organic burger and its packaging; this case study is the first one selected for a more in dept evaluation of the environmental performance of the potential application of PBS packaging materials in the food sector.

2. Methods

2.1. Goal and scope, functional unit and product description

The goal of this study is to quantify the environmental impacts associated with one package of vegetarian burger, comparing its traditional fossil based packaging (baseline solution) with biobased PBS packaging (biobased solution). The attributional modeling approach has been adopted, since the main objectives of the analysis are identification of environmental hotspots and optimization of the product (ecodesign). PBS can be considered a niche product, and therefore is not expected to bring major changes in the market.

The functional unit is 1 package of vegetarian organic burger including its primary (tray, sealing film and cardboard sleeve), secondary (cardboard box) and tertiary (pallet and plastic film) packaging.

One unit of vegetarian burger contains 180 g of food product; its ingredients, in descending order of quantity are: seitan, spinach, potato, onion, sunflower oil, wheat flour, potato flakes and formulation for vegetable broth. Primary packaging is composed by a plastic tray sealed with plastic film and wrapped in a color printed white-lined chipboard sleeve. Secondary packaging is a corrugated (single wall) printed cardboard box and contains three units of primary packaging. Tertiary packaging is composed by a europallet wrapped in polyethylene film; each pallet carries 1584 units of primary packaging. The composition and weight of packaging for the two scenarios (baseline and biobased) are reported in Table 1. In the baseline solution the tray is thermoformed while in the biobased solution the tray is produced through injection molding.

Table 1. Packaging composition for one package of vegetarian burger

Packaging element		**Baseline solution**		**Biobased solution**	
		Material	**Weight (g) [a]**	**Material**	**Weight (g) [a]**
Primary	Tray	PP-EVOH-PP	15.2	80% biobased PBS + 20% talc	22.6
	Sealing film	PET-LDPE-EVOH-LDPE	1.7	Biobased PBS	2.1
	Cardboard sleeve	Coated white lined chipboard	17.5	Coated white lined chipboard	17.5
Secondary	Cardboard box	Corrugated board, single wall	17.0	Corrugated board, single wall	17.0
Tertiary	Europallet	Wood	15.8	Wood	15.8
	Film	LDPE	0.3	LDPE	0.3
Total			67.5		75.3

[a] Weight of each packaging element per functional unit.

The weights of biobased PBS packaging elements (tray and film) are calculated supposing that the volume of material employed is the same as in traditional packaging; the total weight is therefore higher for PBS packaging elements, due to a higher density of the materials.

Biobased PBS is synthesized from glucose in three main steps:

1. Fermentation of glucose to succinic acid
2. Hydrogenation of succinic acid to BDO
3. Polymerization (in molten and solid state) of succinic acid and BDO to PBS

PBS production process has been optimized taking into account technical and LCA feedbacks; further improvement of the technology is still possible, since data refer to the pilot scale. The technology currently implemented involves succinic acid production through yeast fermentation; during fermentation, succinic acid is neutralized with a ammonia and recovered through exchange over ionic resins. Sulfuric acid is used for the regeneration of the ion exchange column and ammonium sulfate is produced as co-product. Biobased BDO derives from succinic acid through catalytic hydrogenation and, finally, polymerization of succinic acid and BDO delivers biobased PBS. High molecular weight PBS is needed for most packaging applications; both melt and solid state polymerization are being applied, along with compounding of the material with additives (e.g. talc, PLA, PBSA, etc.), in order to obtain high molecular weight PBS and PBS grades suitable for target applications.

2.2. System boundaries

Vegetarian burger is produced in Italy, while packaging materials are supposed to be produced in Europe; the Italian and UCTE electricity mixes have been used accordingly. The burger has been supposed to be distributed within Italy, therefore packaging disposal reflects the Italian scenario (ISPRA, 2012). Primary data refer to 2011 and 2012.

System boundaries include production of food ingredients and packaging, raw materials, related transports, manufacturing process of food, distribution to retailers and end of life of packaging. Use phase (retail, storage and cooking) and disposal of food waste are excluded from the analysis. Furthermore, the role of packaging in preserving food quality and safety and its influence on food spoilage and loss have been, for the moment, excluded from the analysis, due to lack of data from specific conservation tests. Infrastructures have been excluded as well, except in case of database processes already containing infrastructures. A synthetic overview of system boundaries is reported in Figure 1.

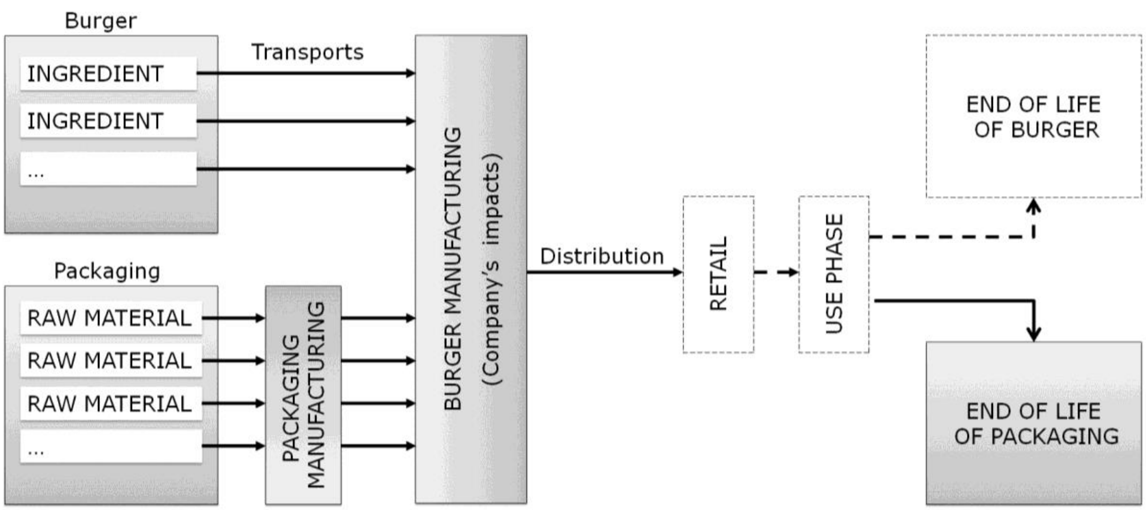

Figure 1.System boundaries. Solid lines: processes included in the system boundaries, dotted lines: processes excluded from the system boundaries.

Concerning biobased PBS granulate, inputs and outputs included in the system boundaries for each processing step are reported in Figure 2. In order to guarantee transparency, atmospheric CO_2 stored in biobased materials has been assessed neutrally, thus no CO_2 credits have been assigned to biobased PBS.

Outputs subject to recycling in the end of life phase (post-consumer), PBS not available for further processing and starch produced during burger's manufacturing, are addressed according to the cut-off approach. Therefore, the environmental impacts of the recycling process are excluded from the system boundaries. Other substances (e.g. water, H_2, N_2) are supposed to be recycled within the PBS granulate manufacturing process.

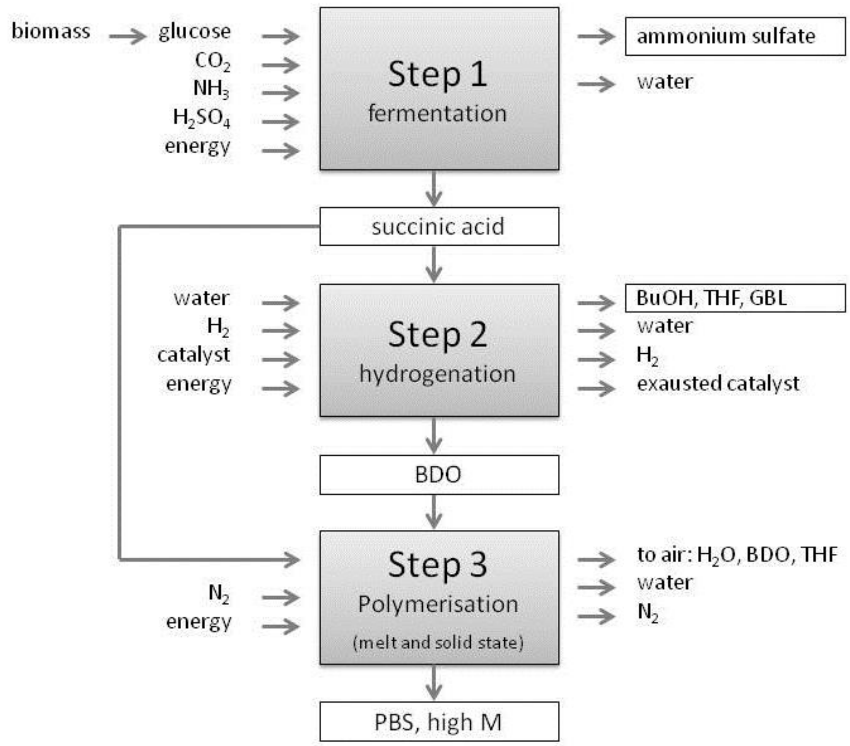

Figure 2. System boundaries: inputs and outputs for biobased PBS production; white boxes: products and co-products

2.3. Data sources and key assumptions

Data on vegetarian burger manufacturing and traditional packaging composition and weight have been provided by the burger's manufacturing company. The main focus of the study is biobased PBS, therefore, while foreground data have been collected from biobased PBS manufacturer, database processes have been used for traditional packaging materials. For the agricultural phase of both food and packaging and for glucose production (from sugar beet), database or literature data have been used (for vegetables: Stoessel et al., 2012). Database processes have been also used for packaging manufacturing (e.g. thermoforming, injection molding, etc.). All database processes derive from the ecoinvent v2.2 database (ecoinvent Centre, 2010). Data not available as foreground data or database information have been approximated; the main assumptions adopted in this study are reported below.

Concerning vegetarian organic burger's ingredients, data referring to organic agriculture have been preferred, while data on Integrated Production (IP) have been used when data on organic farming were not available. LCI data for the production of wheat, potato and sunflower derive from the ecoinvent v2.2 database, while data on vegetables (i.e. spinach, onion) derive from Stoessel et al. (2012). An additional energy consumption has been calculated for frozen ingredients and refrigerated transports, based on Masanet et al. (2008) and Stoessel et al. (2012). It has been assumed that no organic waste, except starch obtained from the manufacturing of seitan, is produced. A transport of 500 km has been assumed for burger distribution.

Data provided by biobased PBS manufacturer are mass balance of main inputs and outputs for the three steps and energy required for the production of succinic acid and melt polymerization. Missing material and energy inputs have been approximated in different ways. Mass of hydrogen and nitrogen required has been calculated based on patent EP 0881203 (Pedersen et al., 2001) and the application of the ideal gas law. Energy required for the hydrogenation step has been supposed to be the same as in the ecoinvent v2.2 process "cyclohexanol, at plant" while cooling energy employed during the polymerization step refers to PET polymerization, as reported by Van Uytvanck et al. (2014). Energy used for solid state polymerization (SSP) refers to an industrial equipment used for the SSP of PET.

In the end of life phase, direct CO_2 and CH_4 emissions deriving from the degradation of plastic materials have been assumed to be of fossil origin for conventional packaging and biogenic for biobased PBS packaging. Waste treatment processes for PBS (e.g. landfilling and incineration) have been created using an ecoinvent tool for the calculation of specific waste datasets (Doka, 2007). According to the carbon neutrality approach, biogenic CO_2 emissions do not contribute to global warming potential.

2.4. Allocation

The recovery of succinic acid from fermentation broth delivers ammonium sulfate as co-product, while the hydrogenation of succinic acid to BDO produces small amounts of n-butanol (4.6% w/w), gamma-butyrolactone (1.6% w/w) and tetrahydofuran (0.3% w/w). In case of BDO, allocation of environmental impacts among co-product has been based on physical relationship (mass), whereas for succinic acid and ammonium sulfate, due to the relatively great amount of ammonium sulfate produced (0.56 kg of ammonium sulfate for each kg of succinic acid), and the remarkable difference in price between the two substances, economic allocation has been applied, assigning 98.6 % of the environmental burden to succinic acid and 1.4% to ammonium sulfate.

For vegetable burger production, being impossible to retrieve data for specific production lines (various food products are produced in the same facility), total energy and water consumption, and waste production of the company (company's impacts) have been allocated to vegetable burger based on its economic value with respect to the overall turnover.

2.5. Life Cycle Impact assessment methods

The methods employed to carry out LCIA are CML baseline v3.01 (Guinée et al., 2002) integrated with the impact category "Land competition" (Guinée et al., 2002) and the method ReCiPe Endpoint H/A v1.10 (Goedkoop et al., 2012). Both methods comprise a set of indicators potentially useful when considering agricultural processes (e.g. euthrophication, human toxicity, ecotoxicity, land use) and allow the normalization of the results. It has been decided to apply one midpoint and one endpoint method, in order to gain a broader overview of the environmental impacts related to the product under study. ReCiPe endpoint, in particular, is applied because it provides a single score indicator, that gives an overall synthetic estimation of the environmental performance of the product. All calculations have been performed with the software SimaPro 8.0.3.

3. Results

The results obtained for the vegetarian burger for the baseline and biobased scenarios are reported in Table 2 and Figure 3 (method CML, characterization). Each impact category has been split into different aspects of the life cycle, in order to identify which processes give the most relevant contribution to the overall environmental impact. The parts of the life cycle selected are: burger ingredients, including transport to burger's production facility; company's impacts, allocated as previously explained; refrigerated distribution; packaging manufacturing, further subdivided into 1^{ry}, 2^{ry} and 3^{ry} packaging and, finally, packaging disposal.

Table 2. Impact assessment of one package of vegetable burger, for 180 grams of food including its packaging. Method CML, characterization.

Impact category	Unit	Biobased solution	Baseline solution
Abiotic depletion	mg Sb eq	0.829	0.676
Abiotic depletion (fossil fuels)	MJ	10.3	8.73
Global warming (GWP100a)	kg CO_2 eq	0.794	0.672
Ozone layer depletion (ODP)	mg CFC-11 eq	0.094	0.061
Human toxicity	kg 1,4-DB eq	0.172	0.111
Fresh water aquatic ecotox.	kg 1,4-DB eq	0.171	0.124
Marine aquatic ecotoxicity	kg 1,4-DB eq	402	262
Terrestrial ecotoxicity	g 1,4-DB eq	1.49	1.22
Photochemical oxidation	g C_2H_4 eq	0.131	0.105
Acidification	g SO_2 eq	5.93	5.24
Eutrophication	g PO^{3-}eq	4.58	4.02
Land competition	m^2a	0.591	0.561

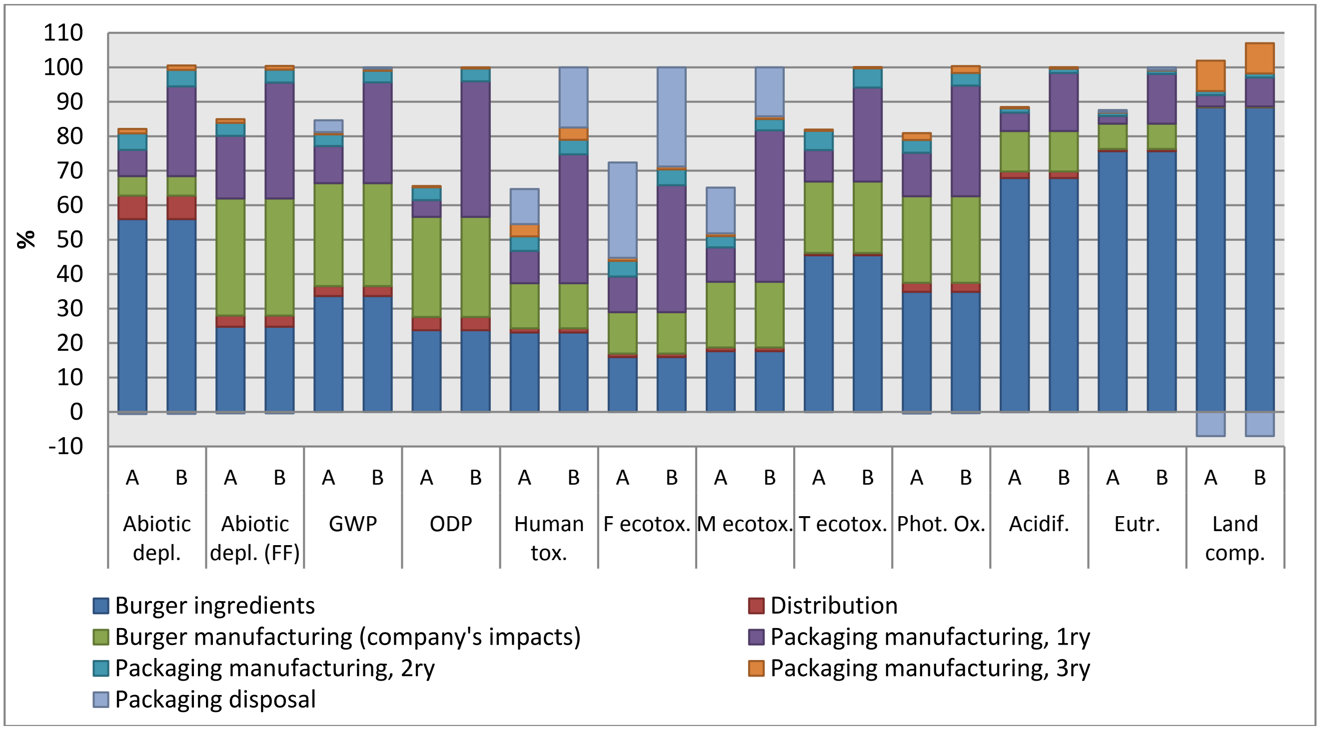

Figure 3. Impact assessment of vegetable burger. A = baseline scenario; B = biobased scenario; method CML, characterization. (FF= Fossil Fuels; ODP = Ozone Layer Depletion; F = Freshwater; M = Marine; T = Terrestrial).

Alternative B (biobased scenario) has higher impacts than alternative A (baseline scenario) for all impact categories considered. The difference between the two scenarios is among 5 % (land competition) and 35 % (human toxicity and marine ecotoxicity). As can be expected, the impact categories that vary most are those with a high contribution of packaging, namely, ozone layer depletion, human toxicity, freshwater ecotoxicity, and marine ecotoxicity. These variations are mainly due to substances emitted in relation to the manufacturing of biobased PBS, for instance, ozone depletion is connected, in particular, to the production of heat, electricity, ammonia and glucose; human toxicity to electricity and liquid carbon dioxide; freshwater and marine ecotoxicity to electricity.

It should be noticed that, since the carbon neutrality approach has been adopted, the LCI model does not account for absorption of atmospheric CO_2 in biobased materials. According to the stoichiometry of the monomer, the CO_2 embedded in biobased PBS is around 2 kg/kg of PBS. Therefore, for each package of spinach burger, approximately 40 g of CO_2 are absorbed from the atmosphere and embedded in biobased plastics (cradle to gate). Some of the carbon absorbed is released in the end of life of packaging; according to the assumptions and waste scenario adopted, biobased packaging disposal releases around 23,1 g of biogenic CO_2 and 16 mg of biogenic CH_4, with a net carbon sequestration of approximately 16.5 g CO_2 per package.

The contribution of food (including distribution) to each impact category varies among 29% (freshwater ecotoxicity, biobased scenario) and 95% (eutrophication, baseline scenario). Concerning the food product, the phases of the life cycle with the highest contributions are burger’s ingredients and company’s impacts, while distribution is less relevant. Burger’s ingredients are particularly important for the impact categories land competition, acidification, eutrophication and terrestrial ecotoxicity, mainly due to the agricultural phase of the life cycle; in case of abiotic depletion, transports give the most relevant contribution.

Concerning packaging, its manufacturing, and in particular the manufacturing of 1^{ry} packaging, has higher impacts than packaging disposal for most impact categories. The negative values in the graph (positive impacts) are due to the reuse of pallets.

The characterization results (not shown) obtained with the method ReCiPe endpoint H/A lead to similar considerations. The ReCiPe characterization results, normalized with respect to the European average, are reported in Figure 4. The impact categories with the highest values after normalization are fossil depletion (FD), agricultural land occupation (ALO), climate change (effects on human health CC/HH and on ecosystems CC/E) and particulate matter formation.

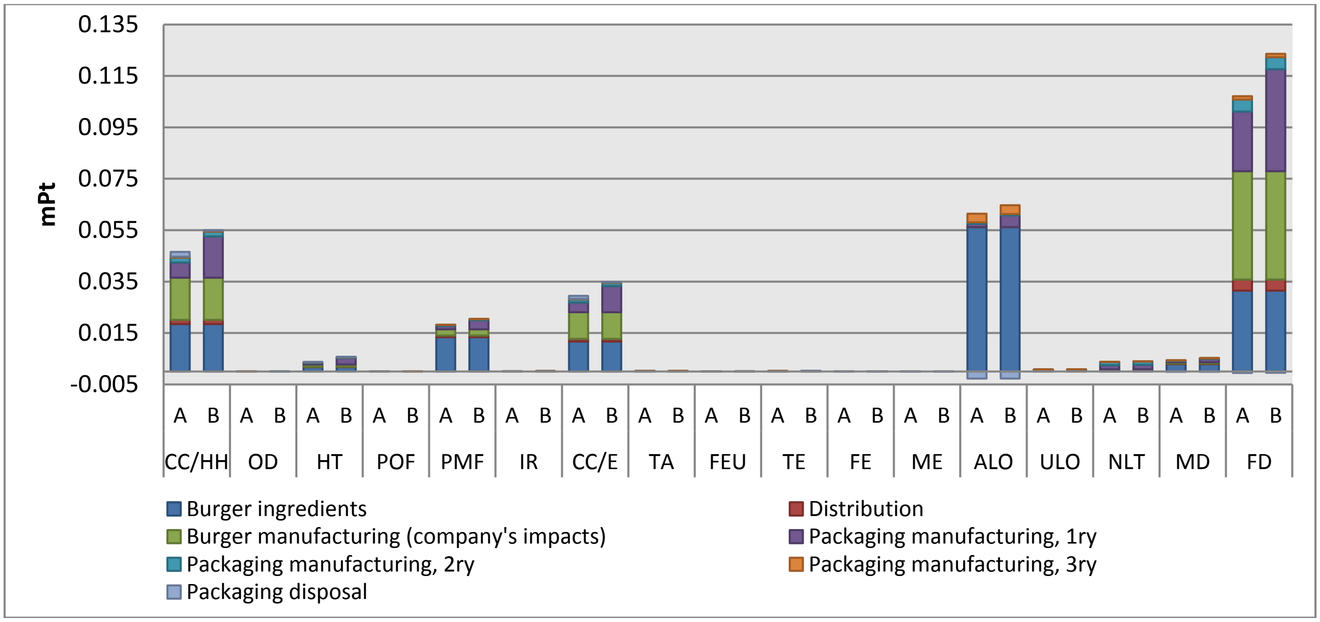

Figure 4. Impact assessment of vegetable burger. A = baseline scenario; B = biobased scenario; method ReCiPe endpoint H/A, normalization.

(CC/HH = Climate Change Human Health; CC/E = Climate Change Ecosystems; OD = Ozone Depletion; TA = Terrestrial Acidification; FEU = Freshwater eutrophication; HT = Human toxicity; POF = Photochemical oxidant formation; PMF = Particulate matter formation; TE = Terrestrial ecotoxicity; FE = Freshwater ecotoxicity; ME = Marine ecotoxicity; IR = Ionising radiation; ALO = Agricultural land occupation; ULO = Urban land occupation; NLT = Natural land transformation; MD = Metal depletion; FD = Fossil depletion).

The ReCiPe single scores values are reported in Figure 5. The biobased solution has a higher single score than the baseline solution (14%), consistently with the results previously outlined for the CML method. The food product (including distribution), has a contribution of 82% in the baseline scenario and 72% in the biobased scenario. For both scenarios, burger's ingredients give the largest contribution to the total score, followed by company's impacts and manufacturing of primary packaging. The impact categories that give the most relevant contributions to the single score indicator in the case of traditional packaging are agricultural land occupation (27%), fossil depletion (24.5%) and climate change (21.4% human health; 13.6% ecosystems), while for the biobased solution the most relevant impact category is agricultural land occupation (25%; mainly due to the agricultural phases of burger and packaging), followed by fossil depletion (24.9%) and climate change (22.2% human health; 14.1% ecosystems).

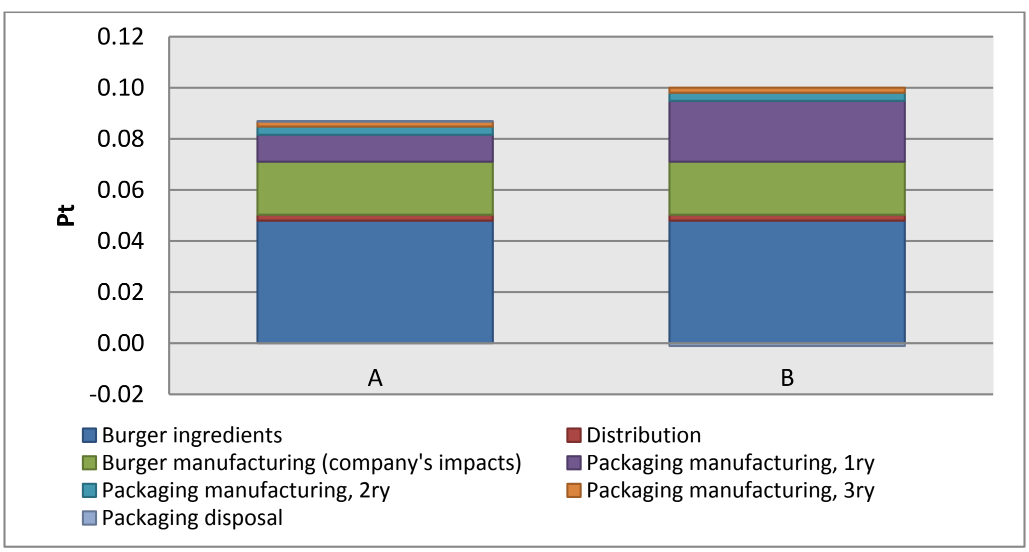

Figure 5. Impact assessment of vegetable burger. A = baseline scenario; B = biobased scenario; method ReCiPe endpoint H/A, single score.

4. Discussion

At the current scale of production (pilot scale) and state of development of the technology, the projected use of novel biobased PBS packaging in comparison to traditional fossil based packaging would increase the environmental impact of the packed vegetable burger. Currently, biobased PBS environmental impacts are higher than the ones of traditional packaging and according to the assumptions adopted a higher amount of biobased material is needed for the same amount of packed food. The environmental impacts of biobased PBS are mainly related to succinic acid production, since succinic acid is used to produce both BDO and biobased PBS. In particular, the technology currently applied for succinic acid synthesis through fermentation has a relatively high energy consumption (32 MJ/kg succinic acid). According to Cok et al. (2014), allocation of environmental burden among succinic acid and ammonium sulfate can be also relevant, economic allocation being the most conservative allocation choice, and therefore, the one with the highest overall environmental impact attributed to biobased PBS.

Another aspect that should not be underestimated in LCA, is the source of database information used; for instance, incomplete LCI (life cycle inventory) datasets could lead to lower environmental impacts than more complete datasets, due to lack of information and not to the actual environmental performance of the product (European Bioplastics, 2012). For example, the ecoinvent v2.2 record for polypropylene (PP is the main material in burger's traditional packaging), is based on inventory data from PlasticsEurope (Hischier R., 2007). These data are highly aggregated and not transparent, and this hampers the identification of possible inconsistencies between LCI and LCIA of PP and PBS.

Moreover, in the case study considered, the difference between the two packaging solutions is more evident than it would be in other applications, since the environmental impact of the vegetable burger (excluding packaging) is relatively low (i.e. GWP 2.9 kg CO_2 eq/kg) and, therefore, the packaging plays a more important role in determining the overall environmental impact.

Whereas traditional plastics are produced through well established and mature technology on industrial scale, entirely biobased PBS is in an early stage of development, and far from full industrial scale, and its environmental performance could improve moving from pilot scale to industrial scale. The main aspects to be considered in order to improve the environmental performance of biobased PBS are the energy required for the synthesis of succinic acid and the yield of the polymerization phase (e.g. reduction of the amount of PBS not available for further processing). In particular, concerning succinic acid production, alternative synthetic routes could be tested. For instance, in the analysis reported by Cok et al. (2014), succinic acid produced through low pH yeast-based fermentation followed by direct crystallization seems to have lower environmental impacts compared to the technology described in the present paper. Concerning polymerization, it seems that the application of a solid state polymerization phase in combination with melt oligomerization could reduce the environmental impact of biobased PBS in comparison to the application of melt polymerization. The applicability of this option is currently being evaluated in order to identify the most efficient solution.

Concerning the whole system (food product + packaging solution), the implementation of some improvement actions could reduce the overall environmental impact. The impact of primary packaging could be reduced by lowering its weight, especially in the case of biobased packaging. The cardboard sleeve, that is present for aesthetic reasons, could be eliminated and substituted, for instance, with a label, or its dimension could be reduced. Other aspects are the reduction of fossil energy employed by burger's manufacturing company and the use of fresh and local ingredients that would decrease the need for refrigerated long distance transports.

One of the main roles of packaging is to preserve food, reducing food spoilage and food waste. This aspect was not yet considered in the present study, but the effects of food loss on the overall environmental impact of the vegetarian burger will be the subject for further evaluations. In particular, some first results from conservation and shelf life studies indicate that PBS packaging could extend the shelf life of some food products and thus presumably contribute to a reduction of food waste and associated environmental impacts. LCA analyses of packaging solutions, therefore, should be focused not only on packaging reduction, but also on enhancing packaging's ability to preserve food and reduce food waste; in some cases, an increase of the environmental impact of packaging could be necessary in order to improve the overall environmental performance of the system, if food loss is included in the system boundaries (Williams and Wikström, 2011).

5. Conclusion

In this case study, LCA has been applied to an existing food product in order to evaluate its environmental impacts and to assess the potential outcomes of the introduction of new biobased PBS packaging materials in the food market. The application of LCA has allowed to identify the critical aspects and improvement options for the vegetarian burger and its packaging, and to provide ecodesign feedback for the development of biobased PBS.

At the current state of development of the technology, the use of biobased PBS instead of conventional fossil based plastics for primary packaging would increase the overall environmental impact of the food product under study. These results should, however, be considered in perspective: while traditional plastics are produced through well established and mature technology, biobased PBS is in an early stage of development, and far from full industrial scale. Moreover, the weight of analyzed PBS packaging is higher than the one of traditional packaging, due to higher density of PBS-based materials. In general, biobased PBS seems to be an interesting material for packaging applications but its environmental performance should be further improved. The increase of the environmental impact of packaging could be justified if this increase could result in a lower rate of food waste, reducing the overall environmental impact of the packaged food product.

6. Acknowledgements

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LCA as a tool for targeted GHG mitigation in Australian cropping systems

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ABSTRACT

Australian agricultural industries contribute approximately 14.6% of net annual national greenhouse gas (GHG) emissions, with N_2O emissions from agricultural soils the second greatest source of these emissions. Given that 25 M ha of land in Australia is cropped, the technical potential for GHG emissions reduction in Australian grain production systems is substantial. The New South Wales Department of Primary Industries (NSW DPI) has developed research capacity in Life Cycle Assessment (LCA) to assess this mitigation potential. In this paper we provide insights into the regionally-specific approach that we are taking, not only to provide credible management options at a grain grower level and ensure that detailed data are available for analysis by participants in the downstream supply chain, but also to provide data which, in an aggregated form, will underpin market access and inform national policy development. We report on initial NSW DPI studies and discuss a new project, funded by the Grains Research and Development Corporation (GRDC), to determine emissions reduction opportunities for each of Australia's agro-ecological zones. Initial studies show total emissions from wheat production in the order of 200 kg CO_2-e per tonne, with values ranging down to 140 kg CO_2-e per tonne. In one study, replacing synthetic nitrogenous fertiliser with biologically fixed N reduced emissions to 33% of prior values. The new project is particularly concerned with developing accurate foreground data by triangulating several sources of published literature (including official statistics) and conducting 'ground-truthing' through panels of regionally-based advisors to increase data specificity. The LCAs and associated mitigation strategies will be underpinned by a median and relevant distribution of values for inputs, practices and yields, with system assumptions clearly documented.

Keywords: cropping, grain production, mitigation, agriculture, emissions

1. Introduction

It can be said with 95% certainty that anthropogenic emissions of the greenhouse gases (GHG) CO_2, N_2O and CH_4 are the primary driver of climate change (IPCC 2013). Australian agriculture industries contributed approximately 79.5 Mt CO_2-e or 14.6% of net annual national greenhouse gas (GHG) emissions in 2010, with N_2O emissions from agricultural soils the second greatest source of emissions after those from livestock methane (Australian Government 2013).With approximately 25 M ha of Australian farming land cropped, the technical potential for GHG emissions reductions in grain production systems in Australia is substantial. All industries have a responsibility to contribute to the reduction of emissions to minimise the impacts of climate change, including agriculture.

The application of Life Cycle Assessment (LCA) to agricultural systems is now relatively widespread (Roy *et al*. 2009) and country-specific life cycle inventories are being developed to increase the applicability of LCAs, e.g. The Australian Agricultural Life Cycle Inventory (Eady *et al*. 2013). Australia has considerable intra- and inter-regional variation in landscape and climate, aspects which determine the most profitable functional production unit. This variation, coupled with methodological issues associated with attributing emissions accurately and accounting for the flow-on effects arising as a consequence of recommended practice changes, creates challenges for LCA practitioners. Also, data gaps exist when attempting to apply published regionally-specific Emissions Factors (EFs). In this paper, we discuss existing studies by the NSW Department of Primary Industries (NSW DPI) and a new project, which we will lead, to explore mitigation options for grain production across Australia.

2. Australian cropping systems

Australia is a relatively large landmass, with climatic zones ranging from wet tropics to temperate semi-arid plains (Williams *et al*. 2002). Australia also has high seasonal climatic variability driven in South-eastern Australia by the El Nino - Southern Oscillation and the Indian Ocean Dipole (Cai *et al*. 2011). Grain cropping is a major agricultural land use occupying approximately 25 Mha, with an estimated annual production of approximately 44 Mt (ABARES 2013). The Grains Research and Development Corporation (GRDC) has determined agro-ecological zones for their industry by using a similar framework to Williams *et al*. (2002), who considered climatic, agronomic and ecological factors. The GRDC's agro-ecological zones have been fine-tuned for relevance to regional grain growing systems and the geographical constraints of those systems (Figure 1), providing suitable biophysical boundaries for LCA studies. The GRDC is one of the primary providers of research funding for the Australian grains industry, raising funds through grower levies and federal government co-contributions.

Grains cropping in Australia primarily occurs where rainfall is < 650 mm, although can occur in high (> 650 mm) rainfall areas. Winter crops such as wheat, barley, oats and canola, with smaller areas of legumes (*e.g.* lupins, peas and vetch), are primarily grown in regions with a typical winter dominant rainfall pattern (*i.e.* southern Western Australia (WA), South Australia (SA), some parts of Victoria and southern New South Wales (NSW)) (ABS 2013). Whilst winter grain crops are also grown in northern NSW and southern Queensland (Qld), summer grain crops, such as chickpeas, sorghum and sunflowers are also grown, to capitalise upon high levels of sub-soil moisture during mid-spring and a greater proportion of rainfall in summer. In tropical zones (*e.g.* WA Ord and Central and Far North Qld) sugarcane dominates, with grain crops grown in rotation as a break crop for disease control. Where irrigation is available, systems are modified to meet market demand, with crop production primarily limited by the capital infrastructure of the enterprise.

Australian agriculture is supported by an effective financial assistance of 0.15% of national gross domestic product (GDP), which is relatively small compared to the European Union (0.73%) and the USA (1%) (OECD 2013). Cropping, sheep and cattle enterprises are aggregated for Australian national economic reporting purposes and receive approximately a third of this assistance (Productivity Commission 2013).

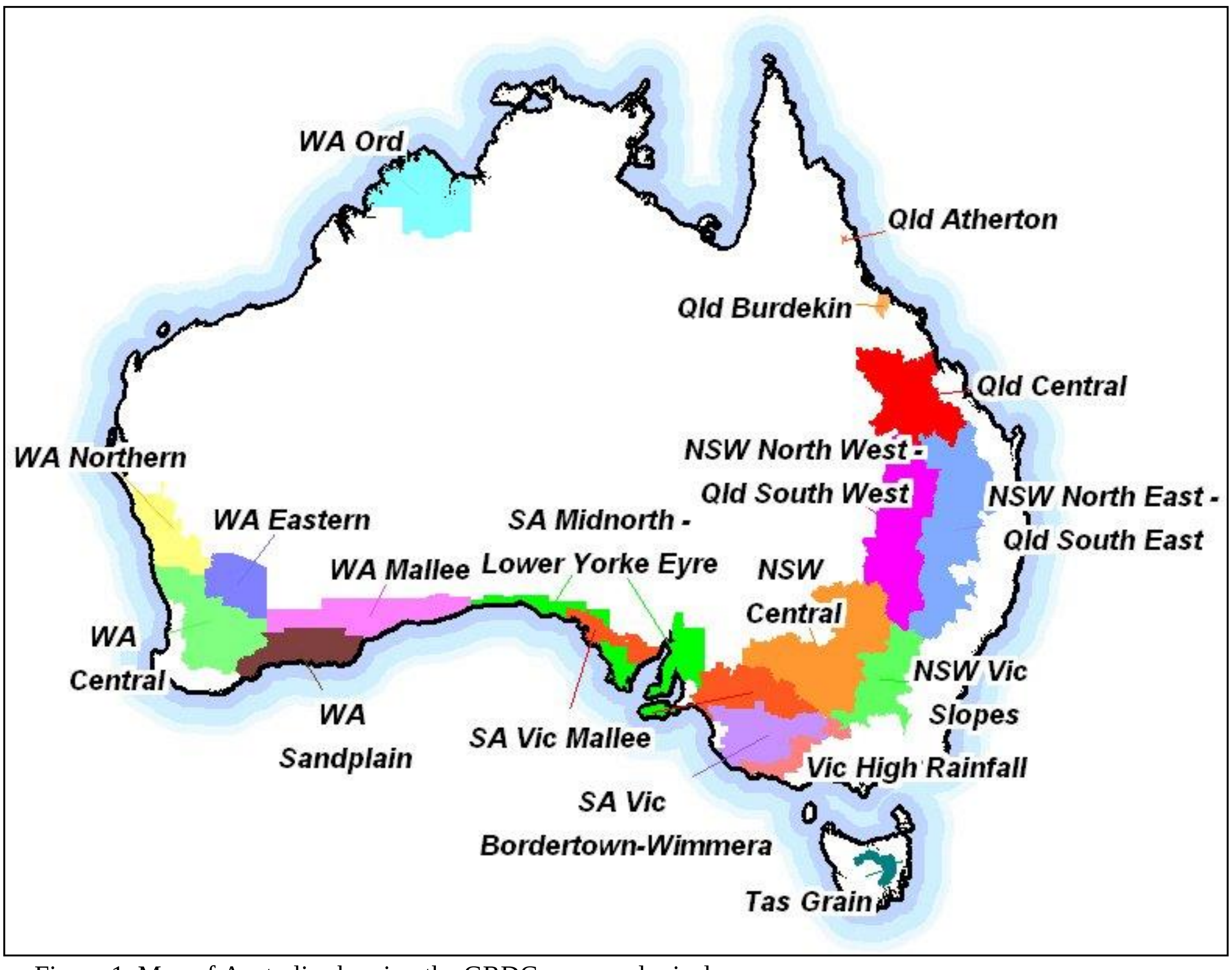

Figure 1. Map of Australia showing the GRDC agro-ecological zones.

The scale of enterprise varies between regions (Table 1) with larger grain growing holdings in WA, compared with other States and Territories of Australia. Scale plays an important role, with larger farms generally more profitable due to economies of scale in purchasing inputs, machinery size and labor costs.

Table 1. Holding area, numbers of holdings and average holding area of cropping properties in the States and Territories of Australia (ABS 2013).

State	Area of holding (ha)	Number of holdings	Average area of holdings (ha)
WA	8 543 799	8 112	1053
SA	4 048 970	9 384	431
NSW	6 931 906	18 069	383
Vic	3 818 961	12 448	306
Qld	2 898 356	11 596	249
Tas	70 581	1 415	49
NT	11 084	238	46

3. Data sources and quality

As with any modeling approach, the quality of input data is a key determinant of the quality of an LCA. In terms of foreground data, LCA research being conducted by NSW DPI takes the approach of triangulating several sources of published literature and then conducting ‘ground-truthing’ with advisors, primarily agronomists and economists. Official statistics, such as from ABS (2013) and ABARE (2013) are included, for attributes for which they are available, along with State-specific official crop statistics reports which provide total tonnages aligned with area sown. However, these data are often not collected at a sufficiently fine scale, do not cover all attributes necessary to conduct LCA and do not universally provide enterprise-specific ranges to enable sensitivity testing. Another key source of data has been annual gross margins, published by NSW DPI, which forecast yield and profit for given inputs and management actions. However, modifications to these data have also been necessary to account for long-term trends.

A new project, funded by GRDC, has a specific budget allowance for panels of regionally-based advisors to fine-tune data and by involving them in the LCA process, they will be well informed to extend findings to their grain grower clients. These advisors have a large client base with approximately half of all grain growers employing advisors in some areas. We are confident that their client data and production knowledge is strongly representative of common practice. Regionally-specific data will be made available for attributes such as the quantities of fertilizer, diesel and herbicide used. The LCAs and associated mitigation strategies will be underpinned by a median and relevant distribution of values for inputs, practices and yields, with system assumptions clearly documented. Sensitivity analysis is then conducted and individual attributes which show significant variation, due to biophysical or management differences, further investigated. For example, we are currently testing the effect of different assumptions about tractor engine capacity, source of inputs, such as herbicides, and use of lime and fertiliser. Whilst larger engines consume more diesel, they enable faster completion of farm practices. Also, biophysical modelling could be beneficial to account for differences in denitrification for a given quantity of N applied, in a specific climatic zone, on different soil types. The degree of interdependence varies between attributes, so we have tested variables independently across a representative range and then discussed linkages.

Emissions from biological processes within agricultural systems have been calculated using EFs obtained preferentially from published field measurements (Schwenke *et al*. 2012), then tier 2 data e.g. from the National Inventory Report (Australian Government 2013) and then tier 1 data (Eggleston *et al*. 2006). The approach of adopting published data where they are more specific than default EFs is consistent with the National Inventory Report (Australian Government 2013). However, Australian coverage of data from published field measurement is not complete, so where emission reduction strategies are being tested through comparative LCA, some standardisation of factors is necessary. Background emissions data about the manufacture and transport of agricultural inputs have been preferentially sourced from the Australasian LCI database (Life Cycle Strategies Pty Ltd 2013) which has been adjusted for Australian conditions, followed by data from Ecoinvent (Hischier *et al*. 2009).

4. The application of LCA to Australian cropping systems

Several LCA studies have been completed for cropping systems in the State of NSW, *e.g.* Brock *et al*. (2012a). Some of these studies and a new funded project are discussed below.

4.1. Aims

The purpose of NSW DPI's LCA studies is to develop an understanding of the various components of emissions profiles for a given functional unit and through comparative LCA, identify practice changes that will target emissions reduction hotspots. For example, the substantial contribution of emissions from both manufacture and use synthetic nitrogenous fertilisers led to testing of the effects of supplying crop N requirements from biologically-fixed N (Brock *et al*. 2012b), resulting in emissions reduction to 33% of prior values.

4.2. Methods

Cradle-to-gate attributional LCAs for wheat production in Central Zone (East) NSW (Brock *et al*. 2012a), and wheat-wheat, canola-wheat and chickpea-wheat rotations in North-East (NE) NSW and North-West (NW) NSW (Muir *et al* 2013), have been developed using SimaPro v7.3.3 (Goedkoop *et al*. 2008). Input data were sourced as per Section 3 above. Pre-farm emissions data were included for the production and transport of all inputs, with the exception of wetting agents, as their contribution to overall emissions profiles were considered to be negligible. On-farm, direct N_2O emissions from the use of synthetic nitrogenous fertilizers were included, as were indirect emissions from the re-deposition of volatilized N as NH_3. N_2O from leaching and runoff of nitrogenous fertilisers were not included as these processes were deemed not to occur, due to soil type and climate, for consistency with the National Inventory Report (NIR) (Australian Government 2013). Direct CO_2 emissions from the hydrolysis of urea were included, as were CO_2 emissions from the dissolution of lime.

Whilst emissions from herbicides used during the pre-crop fallow were included, N_2O and CH_4 emissions from the decomposition of stubble were only included post-crop, as this is when they are attributable to the functional unit. However, residue emissions were excluded where they could result in double-counting with experimental field chamber data, with field-based N_2O EF as high as 0.45%, compared to the current Australian default value of 0.3%. Following our decision rule (consistent with the NIR), to adopt local published values, ahead of NIR data, ahead of IPCC data, it will be necessary to include low EF values obtained from soil with a low clay content in WA, in the future work discussed in Section 4.4 of this paper.

Given that consideration is being given to lowering the Australian default value of 0.3%, for WA or on a wider basis, the 0.3% value is a better estimate for Australia than the 1% IPCC default value. The 0.3% value is in between the 0.45% that has been adopted for some existing studies in NSW and a lower value that will be included for future studies for WA, so will provide a useful standardised value if we wish to hold this attribute constant during comparison of other variables. However, if the next NIR (Australian Government) includes a lower value for Australia, then we will adjust our assumptions accordingly. Stubble burning is not considered common practice in these regions, so was also excluded. All above- and below-ground sequestered carbon (C), remaining after harvest, was considered to be re-emitted through decomposition of crop residues, with soil C flux in a steady state. Emissions from combustion of diesel in tractors for cultivation, sowing and spraying of farm chemicals were included, as were emissions from harvesting and on-farm cartage.

4.3. Results

The emission intensity for one tonne of wheat grown in Central Zone (East) NSW and yielding 3.5 t ha^{-1}, was calculated as 200 kg CO_2-e (Brock *et al*. 2012a). The largest contributors to emissions were the production of fertilizer and lime, and N_2O and CO_2 emissions from these inputs (Figure 2). Opportunities to modify the emissions profile through changes to inputs and practices were discussed, ranging from improved fertiliser management to replacing cultivation with herbicide application.

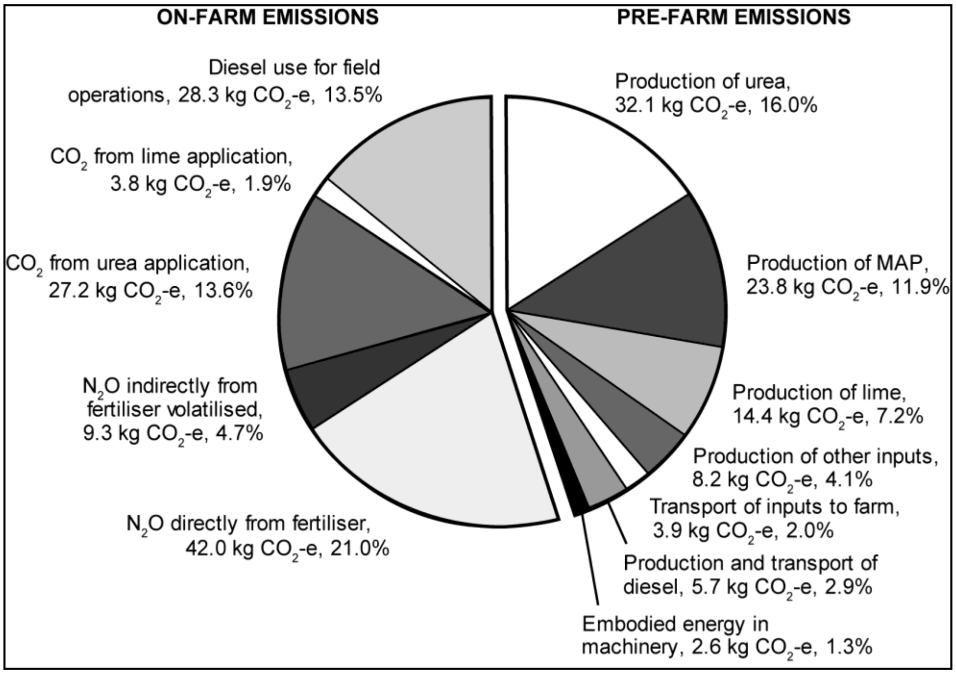

Figure 2. On-farm and pre-farm GHG emissions (CO_2-e) for a tonne of wheat from cradle-to-gate in Central Zone (East) NSW.

Similar studies by NSW DPI, funded by the GRDC (Muir *et al*. 2013), have provided estimated emissions intensities of short fallow wheat production in North East (NE) NSW following wheat (at 218.9 kg CO_2-e), following canola (at 185.3 kg CO_2-e) and following chickpea (at 139.1 kg CO_2-e per tonne of wheat), with 130, 130 and 80 kg N ha^{-1}, respectively. A tonne of wheat grown after chickpeas had an intensity that was approximately 80 kg CO_2-e lower than that grown after wheat. These reductions were primarily driven by the lower quantity of synthetic nitrogenous fertiliser applied (Figure 3). Similar impacts were found for the North West (NW) NSW where a tonne of short fallow wheat grown after chickpeas (at 140.7 kg CO_2-e) was calculated to have an emissions intensity that was approximately 50 kg CO_2-e lower than wheat grown after a wheat crop (at 192.5 kg CO_2-e) (Figure 4). The reduction of emissions intensity of approximately 30 kg CO_2-e/t for wheat following canola for both districts was the result of higher crop yields, given the same fertiliser inputs as wheat following wheat, with a total for one tonne of wheat following canola in NW NSW of 166.1 kg CO_2-e.

When long fallow wheat after sorghum was considered for NE NSW, emissions were found to be 143.4 kg CO_2-e per tonne of wheat (Muir *et al*. 2014a). These emissions are similar to wheat from short fallow after chickpeas but, despite 106 kg N/ha applied, the emissions intensity was much reduced by the higher yield (3.5 t/ha) obtained, compared with 3 t/ha in the short fallow. Emissions were 139 CO_2-e/t for long fallow wheat grown in more arid NW NSW and planted with a total of 32 kg N/ha (Muir *et al*. 2014b). Again a combination of lower N fertiliser application and greater grain yield of 2.4 t/ha reduced the emissions intensity compared with short fallow wheat (1.7-2 t/ha). LCAs are currently being finalised for the pre-crop, grown in rotation with the cereal, and as the product is usually marketed for human consumption, rather than used within the same production system, its emissions profile will be reported separately. This approach is discussed further in Section 5.1 below.

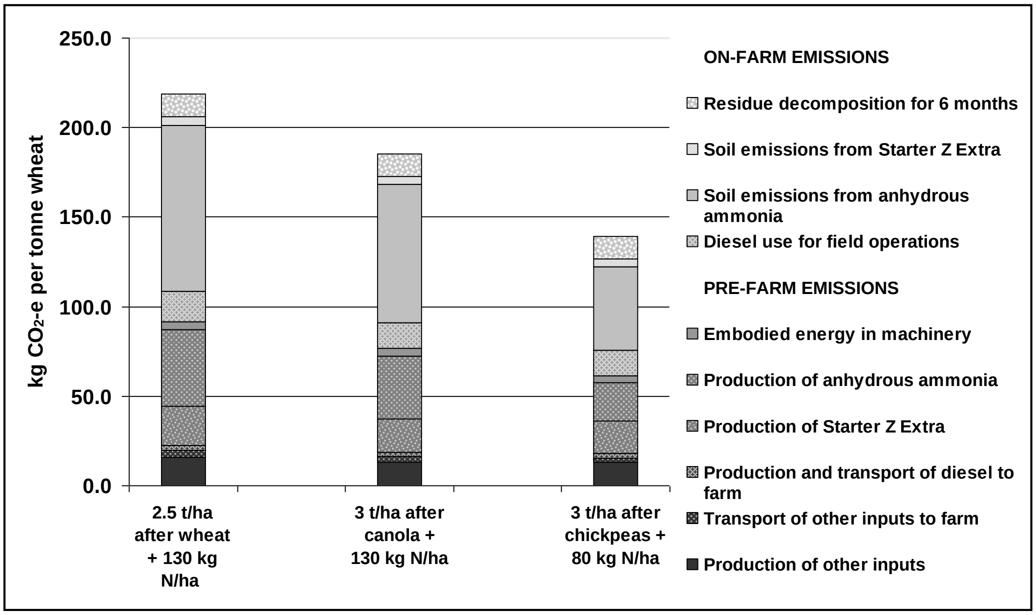

Figure 3. Characterization of GHG emissions for a tonne of short fallow wheat grown in NE NSW in either a wheat-wheat, canola-wheat or chickpea-wheat rotation, with 130, 130 and 80 kg N ha^{-1} applied respectively as combined anhydrous ammonia and Starter Z Extra. Crop yields used in calculations are stated for each rotation.

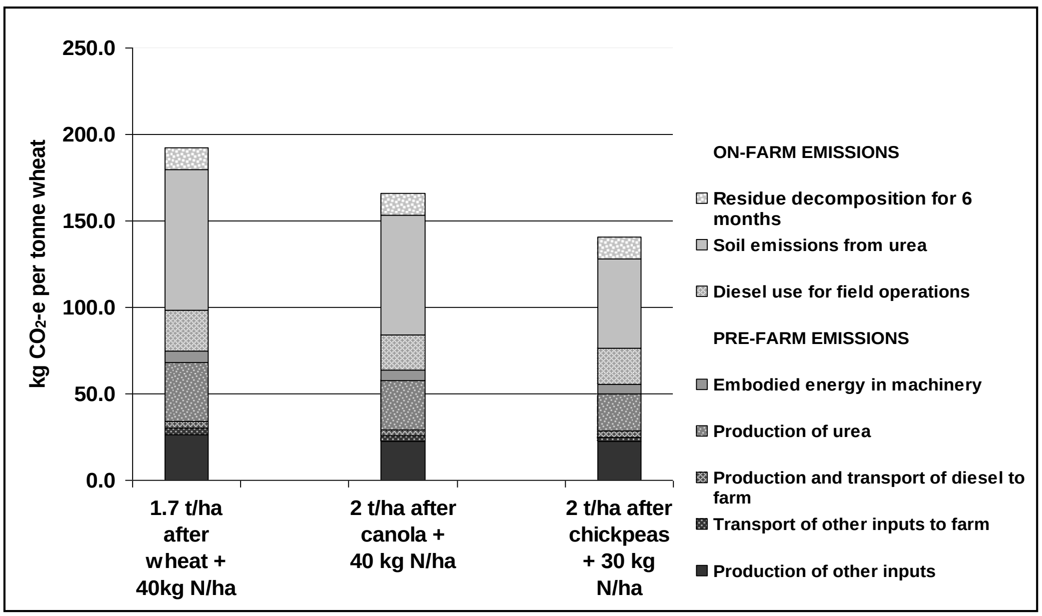

Figure 4. Characterization of GHG emissions for a tonne of short fallow wheat grown in NW NSW in either a wheat-wheat, canola-wheat or chickpea-wheat rotation, with 40, 40 and 30 kg N ha^{-1} as urea respectively. Crop yields used in calculations are stated for each rotation.

4.4. Future work

The GRDC has funded NSW DPI to develop GHG emissions mitigation strategies for the Australian grains industry. This project will involve determining the emissions intensity for a functional unit of the primary grains crop grown in each GRDC agro-ecological zone and for a crop grown in rotation with the primary crop. LCA will be applied to determine emissions intensities and also explore alternative systems and practices that will result in emissions reduction for the same functional unit. It is anticipated that GHG emissions reduction strategies will focus on reducing emissions from the manufacture and use of nitrogenous fertilizers. A possible strategy to achieve this will be by increasing legume rotations into cropping systems to biologically fix N, through rhizobial activity. Emissions reductions may also be achieved by managing excess nitrates in the soil available for denitrification. Possible strategies to achieve this may include splitting fertilizer applications to tailor N availability to crop requirements, using variable rate fertilizer applications to match N application to yield potential and sowing cover crops during the traditional fallow period to 'tie-up' soil nitrates. The effect of different N_2O EFs on this analysis is discussed in Section 5.1 below. The potential to gain minor emissions savings from reductions in fossil fuel use in farm machinery and more energy efficient production of inputs will be examined. The work will also provide a framework for testing future management practice recommendations from an emissions perspective.

Life Cycle Strategies will build non-GHG impact categories, such as land use, water use, eutrophication and ecotoxicity into the LCAs, so that NSW DPI can test for potential unintended consequences of recommended practice changes. The Commonwealth Scientific and Industrial Research Organization (CSIRO) will contribute by developing an impact category to examine the effect of LCA scenarios on soil health. Development of the impact category has commenced and Life Cycle Strategies Pty Ltd will integrate this impact category into the Australian Agricultural Life Cycle Inventory.

Cotton production systems in northern NSW and southern Qld include cereal and legume crops in their rotations and cotton seed is itself a grain. Initial LCA work (Tan *et al*. 2013) has established preliminary emissions profiles for Australian cotton lint and seed production, with separate funding secured by NSW DPI to continue this work. Studies into rice production are also underway (Suenaga *et al*. 2014).

5. Application of results

5.1. On-farm constraints and trade signals

Abovementioned practice change recommendations identified, to date, by NSW DPI researchers include incorporating legumes into cropping rotations, using split fertiliser applications, using nitrogenous fertilisers that have lower emissions potential, growing cover crops during traditional fallow periods, using variable rate fertiliser technology to manage excess nitrates in soils and exploring potential reductions in diesel use. Also, measures to improve productivity tend to provide a net reduction in emissions intensity. Whilst there is some uncertainty about N_2O EFs, with some published factors based on short-term trials, we believe that there are sufficient emerging Australian data from which to test practice change options. NSW DPI has adopted published N_2O EFs, including 0.45% (Schwenke *et al*. 2012) and 0.3% (Australian Government 2013) and will adopt lower published values for WA, where clay content and in some instances rainfall is lower. Current EFs are adequately representative for determining the relative benefit of different practice change options. Values for the absolute benefit can then be refined over time, with emerging data and potential revision to the NIR default values. In the meantime, we will continue to discuss the consequence of choosing different factors, as per Brock *et al*. (2012a) where the sensitivity of the emissions profile to choice of EF was discussed.

In particular, use of biologically-fixed N has the potential to reduce N_2O emissions, primarily through reduced emissions from the manufacture and use of nitrogenous fertilisers. The practical application of these findings by Australian farmers is, however, somewhat constrained. Farmers tend to tailor their systems to manage economic and agronomic risks to maximise profit and current enterprise mix reflects a robust consideration of these factors. Growing legumes (such as chickpeas, peas and lentils) poses different agronomic risks than growing cereals. Sowing time is more critical for legumes, to minimize the risk of frost damage and heat stress, and legumes are more susceptible to fungal diseases. However, adoption may increase with improvements in skill and technology, and will be driven by increased prices for synthetic fertilisers and market pressures for low-emissions intensity products.

The availability of markets is less of a constraint on practice change. Whilst prices for legumes are variable, Australia has established markets for both premium grade legumes for human consumption and legumes for stock feed. The legumes are targeted towards the human consumption market, with the stock feed market providing a lower value market if human consumption quality is not achieved. These markets are able to absorb additional production, increasing the chance of meeting the technical potential for mitigation through the adoption of legumes. However, impacts on price may occur, reducing the level of risk that growers will be willing to accept. Issues of leakage may also arise, from foregone cereal production, requiring augmentation elsewhere and reduced demand for legume production by other nations. Use for stock feed is often not within the same farm boundary but where it is, there will be associated livestock emissions, livestock co-products and displacement of other sources of protein as a feed source. These factors increase the need for consequential LCA. We include grain legumes as an additional product to the cereal, in a two-product system. We have trialled substitution of the legumes to focus on emissions from cereal production, attributing emissions from biological fixed N in lieu of those from synthetic fertilisers, with N treated as a legume co-product. However, it was not possible to be certain about market substitution and we now focus on reporting on the two products, with relative economic benefits and find this approach beneficial, especially when contrasting different production systems in comparative LCA.

Economic risk associated with changing aspects of a system such as those discussed here is generally determined by whether the marginal cost is equal to the marginal benefit. Even without a need to acquire capital assets, the marginal benefit and cost will vary between farm enterprise as a result of holding area, climate and soil type. They will also fluctuate from year to year based on factors such as commodity prices and climatic forecasts. In addition, the status of an Australian GHG emissions reduction policy framework is currently evolving. This is resulting in uncertainty which may constrain practice change, especially when coupled with uncertainty about the potential scope of international import penalties and restrictions between specific countries and requirements for Environmental Product Declarations. LCA is being increasingly adopted to underpin market access, for example for canola importation into European Union markets.

5.2. Policy

As discussed by Plevin *et al*. (2013), attributional LCA has limitations when used for policy development. This is because it does not take into account how a system shock, such as a reduction in demand for the nitrogenous fertilizers, due to increased use of biologically fixed N, would affect the demand for and use of these commodities within the entire agricultural sector under different market conditions. A reduction in demand for nitrogenous fertilizers in cropping enterprises may see fertilizer use, and GHG emissions, increase in other enterprises, such as those associated with livestock production. Alternatively, if the relative advantage of biologically fixed N is reduced, due to lower fertiliser prices and/or expected high cereal prices, then opportunistic farmers may increase fertiliser rates to stimulate yields and/or quality. In these cases, consequential rather than attributional LCA becomes important.

Attributional LCA is useful for policy development within a specific industry sector where internal policies will not necessarily have to consider impacts outside the industry. Further, data generated by attributional LCA models may also be indirectly used to develop government policy by providing input data for computer generated equilibrium or partial equilibrium models which aim to inform policy by modeling the impact of system shocks (*e.g.* C price) on the wider economy. In general terms, attributional LCA provides a useful starting point, especially when conducting paired comparisons and understanding likely emissions reduction ‘hot spots’, but needs to be augmented with consequential assessment when considering broader system boundaries, investigating leakage or making final practice change recommendations.

6. Conclusion

Researchers at NSW DPI have demonstrated the ability to use LCA as a tool to estimate emissions intensity from the production of grains crops under different management actions, particularly at a regionally-relevant scale. The information which has been generated is intended to be useful for farmers who wish to change their management practices to minimise GHG emissions for environmental stewardship or in response to policies, such as emissions trading schemes, where those who do not reduce emissions may be disadvantaged. The information is also intended to play a role to inform decision-making in the post farm-gate supply chain where the environmental credentials of products is increasingly being deemed important. Although this information has only received limited use in policy formulation in Australia, so far, LCA is a tool that has the potential to guide long-term management change in agricultural systems, whether by strengthening market signals or informing direct government intervention.

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Agricultural valorization of organic residues: Operational tool for determining the nitrogen mineral fertilizer equivalent

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ABSTRACT

Organic residues from agriculture and waste and wastewater treatment can be used as organic fertilizers or soil amendments due to their nutrient and organic matter contents. In order to replace mineral fertilizers by organic residues at equivalent nutrient and fertilizer values, the mineral fertilizer equivalent (MFE) of the organic residue must be known. A simple Excel-tool was developed that allowed determination of the nitrogen MFE of organic residues based on their nitrogen content and composition, and nitrogen emissions from field application of the organic residue. Nitrogen field emissions were estimated using simple models and average climate and soil conditions. A global sensitivity analysis revealed that the application method, determining the extent of incorporation into the soil, contributed 66% to the variability of the calculated nitrogen MFE for application of raw pig slurry.

Keywords: agricultural valorization, organic residues, mineral fertilizer equivalent, sensitivity, substitution

1. Introduction

Many organic residues from agriculture as well as waste and wastewater treatment are suitable for an agricultural valorization as organic fertilizer or soil amendment due to their nutrient and organic matter contents. Application of organic fertilizers can replace the use of mineral fertilizers resulting in a financial advantage for the farmer. Furthermore, if organic residues are applied instead of mineral fertilizers and not in addition to, nitrogen (N) and phosphorus (P) emissions will be reduced due to decreased N and P application to farmland. Organic fertilizers have, however, a lower fertilizing value than mineral fertilizers, because nitrogen is present in both mineral and organic forms. Organic nitrogen is only plant available after mineralization by soil microorganisms. For replacing mineral fertilizers by organic residues at equivalent nutrient and fertilizer values, the mineral fertilizer equivalent (MFE) of the organic residues must be known. So far, MFE values reported in literature were typically used when manures or sewage sludge were substituted for mineral fertilizers in life cycle assessment (LCA) studies (e.g. Bayo et al. 2012; De Vries et al. 2012; Johansson et al. 2008; Linderholm et al. 2012) (see Hélias and Brockmann (2014) for a review). Although simple methods for estimating the MFE of organic residues from their composition have been reported (Delin et al. 2012; Water Environment Federation 2005), they have not yet been applied in LCA studies. A decision support tool (MANNER-NPK) for determination of plant available nitrogen from manure has been developed by Nicholson et al. (2013). The decision support tool was designed for the use by farmers to calculate for a specific manure application the plant available nitrogen for the year of application and the following year. Quantification of the plant available nitrogen was based on organic nitrogen mineralization curves and a sophisticated calculation of nitrogen losses considering site specific climate (e.g. temperature, rainfall, wind speed) and soil conditions (e.g. moisture, temperature) (Nicholson et al. 2013).

The aim of this study was to develop a simple Excel-tool designated for the use in LCA studies that allows for determining the nitrogen MFE of organic residues. The nitrogen MFE is calculated based on the nitrogen content and composition of the organic residue, and nitrogen emissions from field application of the organic residues, which are estimated for average climate and soil conditions.

2. Methods

2.1. Nitrogen MFE calculation

We based the nitrogen MFE calculation on the plant available nitrogen (PAN) calculation presented by the Water Environment Federation (2005) for biosolids. But in contrast to Water Environment Federation (2005), we take into account all nitrogen emissions from field application of the residue instead of only NH_3 emissions. In general, the nitrogen MFE calculation is based on the nitrogen content and composition of the residue (ammonium-N, nitrate-N, organic nitrogen), a mineralization rate for organic nitrogen, and estimated nitrogen emissions from field application of the organic residue (Eq. 1).

$$PAN = N_{org} \cdot k_{min} + NH_4\text{-}N + NO_3\text{-}N - N_{emissions} \qquad \text{Eq. 1}$$

where N_{org} is the organic nitrogen content of the fertilizer/amendment [kg $_{Norg}$/kg N_{tot}], $NH_4\text{-}N$ is the ammonium nitrogen content [kg NH_4-N/kg N_{tot}], $NO_3\text{-}N$ is the nitrate nitrogen content [kg NO_3-N/kg N_{tot}], k_{min} is the organic nitrogen mineralization rate factor [-], and $N_{emissions}$ is the sum of all nitrogen emissions from fertilizer application (NH_3, NO_3^-, N_2O and NO_x) [kg N/kg $N_{applied}$]. Organic nitrogen mineralization rate factors for different organic residues are given in Table 1. Nitrogen emissions from fertilizer application are calculated with equations given in section 2.2.

Table 1. Organic mineralization rate factors for different organic residues.

	Mineralization rate factors		
Organic residue	**Year of application**	**Long-term**	**Source**
Dairy cattle or other livestock			
Lagoon water (< 1% DM)	0.40	0.77	Sullivan (2008)
Thin slurry (1 to 5% DM)	0.40	0.77	Sullivan (2008)
Thick slurry (5 to 10% DM)	0.30	0.67	Sullivan (2008)
Solid (> 10% DM)	0.30	0.67	Sullivan (2008)
Separated dairy solids or horse manure[a]	0.10	0.35	Sullivan (2008)
Compost	0.10	0.35	Sullivan (2008)
Solid poultry (> 10% DM)	0.50	0.87	Sullivan (2008)
Activated sludge (biosolids)	0.40	0.75	Water Environment Federation (2005)
Supernatant[b]	0.40	0.75	Water Environment Federation (2005)

[a] dairy solids from a mechanical separator.
[b] from activated sludge processes.

2.2. Calculating nitrogen emissions from field application

Ammonia emissions from field application of organic fertilizers and amendments were calculated with the AGRAMMON model (Nemecek and Schnetzer 2012), which takes into account application method, time of incorporation after application, and weather conditions during application, among others.

$$NH_3 - N = TAN \cdot \left(EF + c_{app}\right) \cdot c_x \qquad \text{Eq. 2}$$

where $NH_3\text{-}N$ is the emission of ammonia [kg NH_3-N], *TAN* is the total ammonium nitrogen [kg N], *EF* is the emission factor [%TAN/100], c_{app} is a correction factor influencing the emission factor (applies only for liquid manure; [-]), and c_x is a correction factor taking into account impacts of application method and time [-]. Correction factors were taken from Agrammon Group (2013b). Emission factors for a range of organic residues are given in Table 2.

Table 2. Ammonia emission factors for different organic residues.

Organic residue	Emission factor [%TAN]	Source
Cattle manure, liquid	50	Agrammon Group (2013b)
Cattle manure, solid	80	Agrammon Group (2013b)
Pig manure, liquid	35	Agrammon Group (2013b)
Pig manure, solid	60	Agrammon Group (2013b)
Poultry manure, liquid and solid	40	Agrammon Group (2013b)
Horse, sheep, goat manure	70	Agrammon Group (2013b)
Compost, solid digestate	80	Agrammon Group (2013b)
Liquid digestate	60	Agrammon Group (2013a)
Activated sludge (biosolids)	50	Water Environment Federation (2005)
Supernatant[a]	50	assumed to be the same as for biosolids

[a] from activated sludge processes.

Nitrate emissions were estimated with the Smaling (1993) model, proposed by Roy et al. (2003), because the model enables to discriminate between nitrate emissions originating from organic residue application and background nitrate emissions from soil organic matter nitrogen.

$$NO_3 - N = \left(N_{min,soil} + N_{fert}\right) \cdot \left(0.021 \cdot P_{prec+irr} - 3.9\right)/100, \quad \text{for } c < 35\% \qquad \text{Eq. 3}$$
$$NO_3 - N = \left(N_{min,soil} + N_{fert}\right) \cdot \left(0.014 \cdot P_{prec+irr} + 0.71\right)/100, \quad \text{for } 35\% < c < 55\% \qquad \text{Eq. 4}$$
$$NO_3 - N = \left(N_{min,soil} + N_{fert}\right) \cdot \left(0.0071 \cdot P_{prec+irr} + 5.4\right)/100, \quad \text{for } c > 55\% \qquad \text{Eq. 5}$$

NO_3-N is the amount of N leached to groundwater [kg N], $N_{min,soil}$ is the amount of mineralized N in the upper 20 cm of the soil [kg N], N_{fert} is the amount of N applied with organic residues [kg N], $P_{prec+irr}$ is the sum of precipitation and irrigation [mm/year], and c is the clay content of the soil [%]. To obtain only nitrate emissions directly related to organic residue application, $N_{min,soil}$ was set to zero.

Nitrous oxide emissions were calculated with the IPCC method (IPCC 2006) following the equation given in Nemecek and Schnetzer (2012).

$$N_2O = 44/28 \cdot \left(0.01 \cdot \left(N_{fert} + N_{cr}\right) + 0.01 \cdot NH_3 - N + 0.0075 \cdot NO_3 - N\right) \qquad \text{Eq. 6}$$

where N_2O is the emission of nitrous oxide [kg N_2O], N_{cr} is the nitrogen contained in crop residues [kg N], and *NH_3-N* and *NO_3-N* are the ammonia and nitrate emissions calculated with equations given above.

Nitrogen oxide emissions were estimated according to Nemecek and Schnetzer (2012) based on nitrous oxide emissions. NO_x and N_2O are given in kg NO_x and kg N_2O, respectively.

$$NO_x = 0.21 \cdot N_2O \qquad \text{Eq. 7}$$

2.3. Determination of mean soil and climate conditions

Estimation of nitrate emissions from organic residue application required information on annual precipitation and irrigation, as well as on the clay content of the soil. Mean climate and soil conditions have been determined for five European countries: Denmark, France, Germany, The Netherlands, and Poland (Table 3).

Table 3. Mean soil and climate conditions used.

Country	Precipitation (mm/year)	Irrigation[a] (mm/year)	Agricultural surface area with clay content < 35%	Agricultural surface area with clay content 35-55%	Agricultural surface area with clay content > 55%
Denmark	718[b]	1.7[d]	100.0%[i]	0.0%[i]	0.0%[i]
France	700[b]	15.0[e]	93.3%[j]	6.7%[j]	0.0%[j]
Germany	757[b]	1.6[f]	100.0%[k]	0.0%[k]	0.0%[k]
Netherlands	854[b]	7.4[g]	92.5%[l]	7.5%[l]	0.0%[l]
Poland	595[c]	0.6[h]	99.0%[m]	1.0%[m]	0.0%[m]

[a] Irrigation per total agricultural surface area.
[b] ECA&D (2013)
[c] The Word Bank (2011)
[d] Eurostat (2013) and Statistics Denmark (2014)
[e] Commissariat Général au Développement Durable (2012) and Agreste (2013)
[f] Statistisches Bundesamt (2010) and Statistisches Bundesamt (2012)
[g] CBS et al. (2013) and Eurostat (2014)
[h] Central Statistical Office (2011)
[i] Adhikari et al. (2013)
[j] GIS Sol (2013)
[k] Düwel et al. (2007) and Bundesanstalt für Geowissenschaften und Rohstoffe (2007)
[l] WUR-Alterra (2006)
[m] FAO (2003)

2.4. Sensitivity analysis

We carried out a global sensitivity analysis to evaluate the impact of different parameters on the calculated nitrogen MFE value of a selected organic residue (raw pig slurry). Parameters included in the analysis and their ranges and discrete values, respectively, are given in Table 4. Based on the clay contents of agricultural soils in the five European countries considered (see Table 3), it was assumed that agricultural soils do not have a clay content > 55%. The variance-based extended Fourier amplitude sensitivity test (extended FAST) was applied, which allows for estimating the contribution of individual input parameters to the variance of the output variable (Saltelli et al. 2000). First-order sensitivity indices (S_i) provide the fractional contribution of individual parameters to the variance of the output. Total sensitivity indices (S_{Ti}) estimate the contribution of individual parameters including interaction effects between parameters, and are defined as the sum of sensitivity indices involving parameter *i*. Sample generation and calculation of sensitivity indices was carried out using Simlab 2.2 (European Commission and IPSC 2004).

Table 4. Parameter values and ranges used in the global sensitivity analysis

Parameter	Symbol	Values/range	Distribution	Source
Application method (correction factor)	c_{Method}	0.2; 0.3; 0.5; 0.7; 1.0	Discrete	Agrammon Group (2013b)
Season of application (correction factor)	c_{Season}	0.95; 1.00; 1.15	Discrete	Agrammon Group (2013b)
Daytime of application (correction factor)	$c_{Daytime}$	0.8; 1.0	Discrete	Agrammon Group (2013b)
Application on unseasonably warm day (correction factor)	c_{Warm_day}	0.96; 0.98; 1.00; 1.05	Discrete	Agrammon Group (2013b)
NH_3 emission factor (for raw pig slurry)	EF	0.35-0.75	Uniform	this study
Mineralization rate factor for organic nitrogen	k_{min}	0.4-0.8	Uniform	this study (first-year vs. long-term mineralization rate)
Annual precipitation and irrigation [mm/year]	$P_{prec+irr}$	600-850	Uniform	this study
Agricultural surface area with clay content < 35%	clay1	0-1	Uniform	this study
Agricultural surface are with clay content 35-55%	clay2	1-clay1	Relation	this study

3. Results

The developed Excel-tool requires as input information on nitrogen content and composition of the organic residue, type/nature of organic residue, application method and time of field application, and geographical location (for climate and soil conditions). First, field emissions of NH_3, NO_3^-, N_2O and NO_x during organic residue application are calculated. Then, the nitrogen MFE for the first year and for the long-term are calculated. Figure 1 shows nitrogen MFEs calculated for field application of raw pig slurry with 4 kg N_{tot}/m^3, 2.5 kg NH_4-N/m^3, 1.5 kg N_{org}/m^3, different application methods and different geographical locations. Nitrogen MFEs calculated for Danish and French climate and soil conditions were very similar. Total annual precipitation and irrigation differed only slightly (5 mm/year) between Denmark and France, but in contrast to Denmark, 6.7% of the French agricultural surface area had a clay content between 35 and 55% (Table 3). The lowest nitrogen MFEs were calculated for Dutch soil and climate conditions, while the highest MFEs were obtained for Poland. Poland had the lowest annual precipitation and irrigation, whereas the Netherlands had the highest annual precipitation and irrigation. For the same application method, the MFE increased by about 30% when the long-term mineralization rate factor for organic nitrogen was used instead of the mineralization rate factor for the year of application. For the same mineralization rate factor, nitrogen MFEs increased for application methods that allowed for better incorporation of the applied fertilizer into the soil, because burying the applied organic residue decreases NH_3 volatilization.

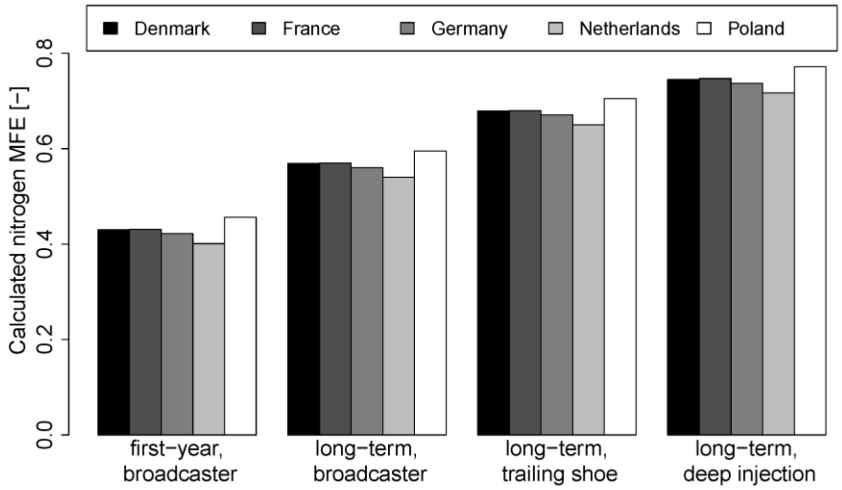

Figure 1. Calculated nitrogen MFE values for different application methods and geographical locations

Nitrogen MFE values calculated for the same application method did not vary much between the selected countries. The variations were mainly due to differences in annual precipitation and irrigation, as the soil types (clay content) were very similar for the different countries. To evaluate the impact of more extreme soil and climate conditions not covered by the selected countries, the clay content of the soil was assumed to be either <35%, between 35% and 55%, or >55%, and the annual precipitation and irrigation was assumed to be 250, 500, 750 or 1000 mm/year. First-year nitrogen MFE values calculated for broadcaster application and different soil and climate conditions are given in Figure 2. Nitrogen MFE values differed most for varying annual precipitation and irrigation for a clay content of the soil <35%. The differences in calculated nitrogen MFE values due to varying precipitation and irrigation decreased with increasing clay content of the soil. Independent of the clay content of the soil, nitrogen MFE values were higher for lower precipitation and irrigation due to lower nitrogen emissions. For low annual precipitation and irrigation (250 mm/year), nitrogen MFE values were higher for low clay contents, while for high annual precipitation and irrigation (1000 mm/year), nitrogen MFE values were higher for higher clay contents.

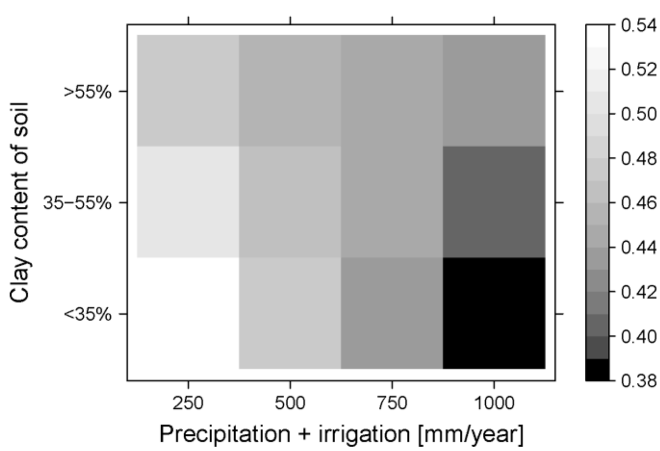

Figure 2. Calculated first-year nitrogen MFE values for broadcast application for different clay contents of soil and different annual precipitation + irrigation.

Calculated nitrogen MFEs presented above showed that application method, mineralization rate factor for organic nitrogen, amount of annual precipitation and irrigation, and clay content of the soil affected the calculated nitrogen MFEs. In order to quantify the impact of the different parameters on the variance of the MFE, a global sensitivity analysis using the extended FAST method was carried out. Figure 3 a) shows the large variability in the nitrogen MFE as a result of the variability in the model parameters. The contribution of individual parameters to the variance of the nitrogen MFE is presented in Figure 3 b). The type of application method chosen had clearly the largest impact on the calculated MFE followed by the NH_3 emission factor (*EF*) and the mineralization rate for organic carbon (k_{min}). The variability in the application method contributed 66.4% to the uncertainty in the calculated MFE based on the first-order sensitivity index and 73.1% based on the total sensitivity index. Contributions of the NH_3 emission factor and the mineralization rate factor for organic nitrogen to the variance of the MFE were very similar with 10.3% and 11.1% (first-order sensitivity index), respectively. All other parameters had only a minor impact (< 5%) on the calculated MFE.

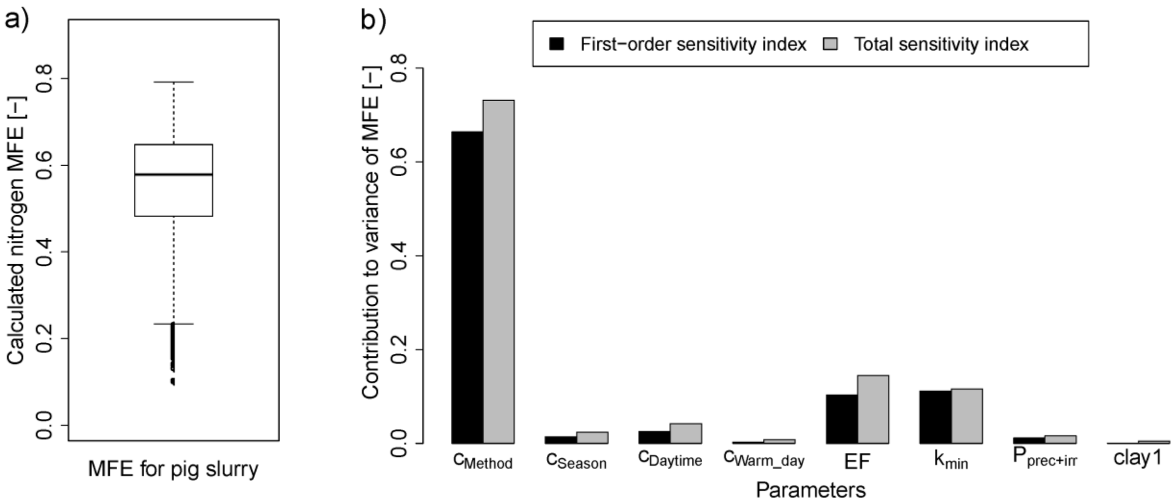

Figure 3. Uncertainty and sensitivity analysis results: a) Box-Whisker plot of the calculated nitrogen MFEs for raw pig slurry. The boundaries of the box mark the 25th and 75th percentile, and the line within the box marks the median. Whiskers above and below the box indicate the furthest points that are within 1.5 times the interquartile range (box length) from the end of the box; b) Contribution of the individual parameters to the variance of the calculated nitrogen MFE based on the first-order and total sensitivity indices calculated with the extended FAST method.

4. Discussion

The developed Excel-tool was based on simple models describing emissions from field application of organic residues. The model for estimating NH_3 volatilization losses accounted only indirectly for climate conditions through different correction factors. In contrast, Nicholson et al. (2013) considered in their decision support tool the impact of soil moisture content, wind speed, land use, and rainfall after spreading on the volatilization of NH_3. But it is difficult to say how much climatic conditions influence NH_3 emissions. Webb et al. (2012) showed that the measuring method and the incorporation of manure (vs. no incorporation) had significant effects on NH_3 emissions. The latter agrees with our results from the sensitivity analysis. When fixing the application method (or the extent of fertilizer incorporation) to broadcaster application without incorporation, the variability in the ammonia emission factor had the largest impact (55%) on the variance of the nitrogen MFE (data not shown). Thus, reducing the variability of the ammonia emission factor by determining, for example, a standard measurement method for ammonia field emissions, would significantly decrease the variability in the estimated nitrogen MFE.

In the calculation of the nitrogen MFE, we did not differentiate between different soil types when applying organic nitrogen mineralization rates. Shah et al. (2013) observed, however, that mineralization rates of organic nitrogen from organic residues vary widely with soil types. But the sensitivity analysis carried out in this study showed that the variation of the mineralization rate factor for organic nitrogen in the range of 0.4-0.8 contributed only 11% to the variation in the nitrogen MFE. Even for a fixed application method (broadcaster application), the contribution of k_{min} to the variance of the nitrogen MFE increased only slightly to 16% (data not shown). As a consequence, reducing the variability in k_{min} by considering the soil type would result only in a minor decrease in the variability of the nitrogen MFE. Thus, the simplification of not differentiating between soils considered to be acceptable.

Our goal was to develop a tool that is easy to apply and does not require many inputs. We think that using simple models and average climate and soil conditions for estimating the nitrogen MFE is reasonable for LCA studies, because LCA studies are often based on a regional or national level. Nonetheless, applying site specific soil and climate conditions and mineralization rates for organic nitrogen may reduce the uncertainty of the calculated nitrogen MFE for a specific case.

5. Conclusion

An Excel-tool was developed that allows for fast and easy determination of the nitrogen MFE of organic residues valorized as organic fertilizers or soil amendments. The nitrogen MFE is needed for calculating the amount of substituted mineral fertilizers. The tool also calculated nitrogen field emissions from organic residue application. The nitrogen MFE was estimated using simple models and average climate and soil conditions for estimation of nitrogen field emissions. A global sensitivity analysis using the extended FAST method was used to quantify the contribution of individual parameters to the uncertainty in the calculated nitrogen MFE. For the case of raw pig slurry application, the type of application method selected (determining the extent of incorporation into the soil) contributed most (66.4%) to the variability of the nitrogen MFE, followed by the NH_3 emission factor and the mineralization rate for organic carbon. It is, therefore, essential to determine the nitrogen MFE for a specific application method to reduce the uncertainty in the calculated nitrogen MFE and with it the uncertainty in the amount of replaced mineral fertilizers. The Excel-tool is available from the authors upon request.

6. Acknowledgements

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Developing Environmental Footprint, Cost, and Nutrient Database of the US Animal Feed Ingredients

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ABSTRACT
The main objective of the US Animal Feed Database was to develop a robust nutrient, economic, and environmental footprint database in one location. The database will be integrated into the US Pig Production Environmental Footprint Calculator (PPEFC). This paper provides an overview of 180 animal feed prices, climate change impacts, water consumption, and land use inventory. The environmental footprint data was based on life cycle assessment (LCA) ReCiPe impact method. The cost and environmental footprint results were analyzed in the context of a single baseline swine diet to evaluate the effect cost and environmental footprint of each feed ingredient based on its inclusion rate. The database potential is seen within the PPEFC calculator where the farmers would be able to model diets beyond the baseline diets. In addition, this database will be used to project least cost and environmental footprint diet formulations which would also be integrated within the PPEFC calculator.

Keywords: animal feed database, environmental footprint calculator, swine diet, feed cost.

1. Introduction

Pork producers are committed to protecting their resources (water, land, and air) and are searching for production strategies of minimum cost to comply with an array of federal, state, and local environmental regulations. In 2011, the US National Pork Board, supported by the USDA Climate Change Mitigation and Adaptation in Agriculture grant, launched a project to develop a science-based decision tool called Pig Production Environmental Footprint Calculator (PPEFC). The PPEFC has the ability to calculate impact to climate change, costs, land use, and water consumption across the pork production chain, including feed formulation and crop production. The combined analysis of all of these factors allows identification of potential ecologically and economically feasible production practices for pork producers.

One of the goals of the PPEFC project is developing an environmental footprint, cost, and nutrient database of the US animal feed ingredients and integrating it with the calculator. The PPEFC is built upon cradle-to-farm-gate life-cycle assessment (LCA) of pork production combined with the US National Resource Council (NRC) swine nutrient requirements models (NRC 2012), farm operation inputs, and animal feed database. Farm operation inputs include: barn characteristic, utilities, manure management, dead animal disposal, and farm operation costs. The animal feeds database includes: environmental footprint, feed prices, and nutrient composition, as seen in Figure 1.

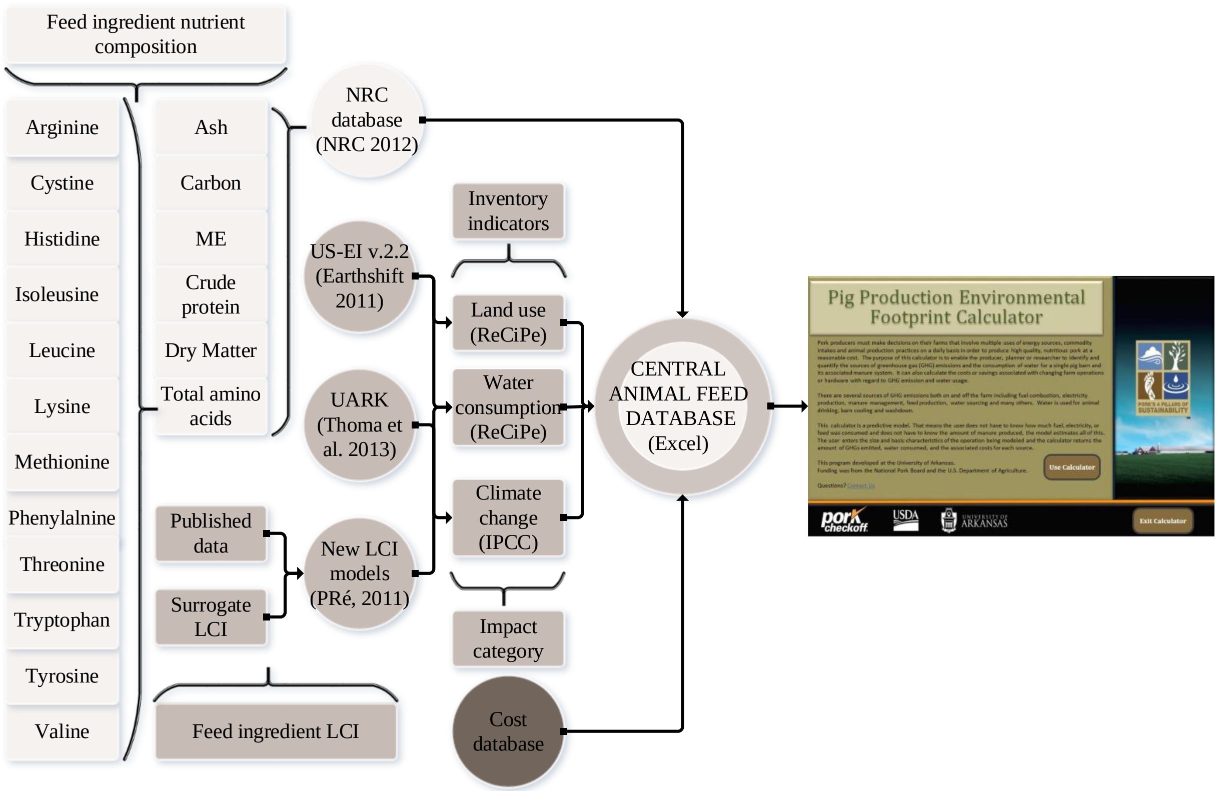

Figure 1. Animal feed ingredient nutrient content, cost, land use, water consumption, and climate change impact database structure and integration into pig production environmental footprint calculator.

This paper analyzes environmental footprint and cost data for swine feed ingredients. Feed ingredients used in swine diet are numerous and of various origins: production of grains and protein crops specifically to feed livestock (e.g., corn grain); by-product feeds from the production of human food and biofuel (e.g., corn meal and Dried Distillers Grains with Solubles (DDGS)); and minerals, vitamins, and other additives from chemical production. According to the NRC (2012), there are more than 180 feeds that could be part of the swine diet in the US. In addition, the US animal feed database contains ingredients that are part of pork production management practices: phytase, ractopamine, and antibiotics.

2. Methods

2.1. Data sources of animal feed nutrient, cost, and environmental footprint

The system boundaries for collection of animal feed ingredients are defined in Figure 2. The main animal feed nutrient content data were obtained from the NRC (2012). Feed ingredient prices were collected from various feed market data sources, including Feedstuffs (2014), a weekly newspaper for agribusiness as well as from feed mills, pork industry representative and firms engaged in the production of feed additives. The animal feed cost values were provided from the Department of Agricultural Economics & Agribusiness at the University of Arkansas (Popp 2014). The main environmental footprint data (40%) for swine feeds was obtained from the US agricultural and product LCA models built in SimaPro 7.3.3 (PRé Consultants 2011). These models are a result of several years of work on fluid milk, poultry, cheese, peanuts, and swine LCA projects at the University of Arkansas (Van Loo et al. 2011; Mccarty et al. 2012; Thoma et al. 2013b; Kim et al. 2013; Adom et al. 2013; Nutter et al. 2013; Thoma et al. 2013a; Thoma et al. 2013c; Thoma et al. 2013d). Standard LCI databases were used (4%): US-EI v2.2 (EarthShift 2011) and US Input-Output database (Mongelli et al. 2005). Direct LCIA results from published papers (Tan et al.; Nielsen and Wenzel 2007; Mosnier et al. 2011) represent less than 10% of data sources. Approximately 7% of data uses surrogate LCI which are used to bridge data gaps for animal feed ingredients.

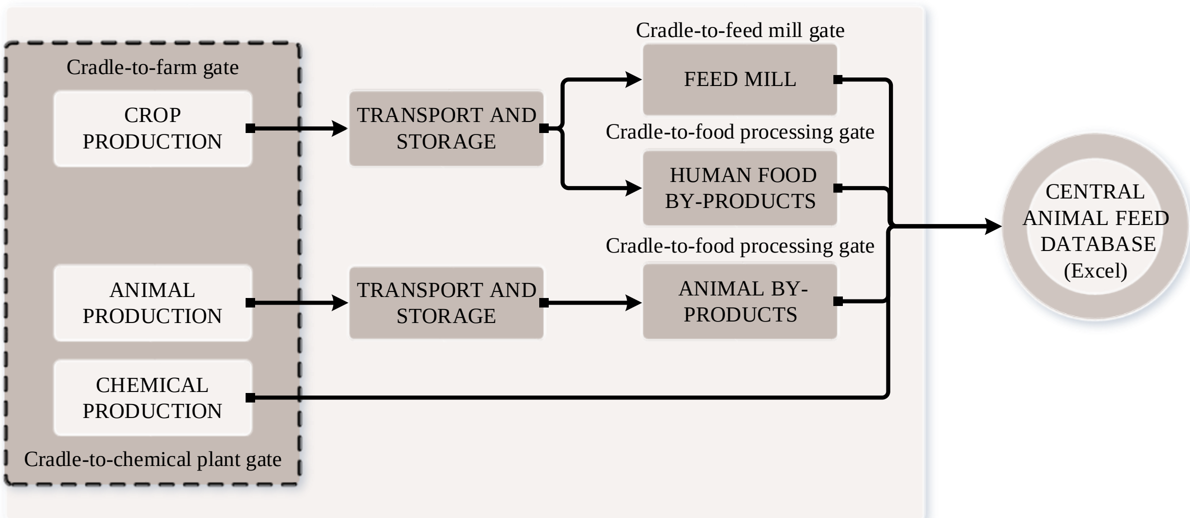

Figure 2. System boundary for feed ingredient production.

2.2. Limitations of the animal feed database

Potential limitations of the Animal Feed Database include the use of different LCI databases, direct LCIA results, multiple, surrogate, and incomplete models. The US-EI v2. 2 (EarthShift 2011) represents the largest database source of swine feed nutrients. The US-EI database (EarthShift 2011) is used to bridge the current gap in the USLCI database (NREL 2012) and applies US electrical conditions to the Ecoinvent database (Ecoinvent, 2010). Input-output databases are based on national economic and environmental statistics. The main disadvantage of this approach is that processes are relatively aggregated and models don’t include water consumption. Sometimes, for similar feeds the same models were used multiple times based on product group. For example, the animal and marine fats and oils model (Mongelli et al. 2005) can be used for animal oils, grease, tallow, bones, fish, and animal meal processing plants. Animal and marine fats and oils model is an input-output model based on market value of the product. The distinction between the environmental footprint of animal and fish processing by-products stems from individual feed prices. Multiple, surrogate, and incomplete models may lead to overestimation or underestimation of the results depending on the relative environmental footprint and percent inclusion of feed in a swine diet. For example, feed ingredient of different origin was used to represent one or multiple feeds. It is important to note that the percent of current use of these feeds in practice is assumed low because production and availability of those feeds in the US is not high. Also, average soybean meal production LCI was used to represent different soybean meal production practices. Direct LCIA results are limited in the number of environmental footprint categories presented (e.g., only GHG emissions available) and published results may have used different LCIA method to calculate the environmental footprint. The results presented in this paper are not final, as the project is still ongoing. With new databases such as ecoinvent 3 database (Weidema et al. 2013) and Agri-footprint v1.0 database (Blonk Consultants 2014) we the US feed database we should be able to avoid the use direct LCIA results.

2.3. Environmental footprint method

The environmental footprint in this context relates to: climate change impact based on 100 year Global Warming Potential (GWP) (Solomon et al. 2007) and inventories of water consumption and land use (Goedkoop et al. 2009). Water consumption and land use denote the ecological footprint of key resource consumption indicators. According to Goedkoop et al. (2009), the consumed water (m^3) refers to a fraction of water that is evaporated or exported to another location from the watershed or released to a different watershed. Land use (m^2a) is described as an inventory of agricultural and urban land occupation (Goedkoop et al. 2009). Climate change impact also deals with resource use (e.g., fossil fuel), but it’s focused strictly on the greenhouse gas (GHG) emissions. The majority of feeds have been converted from external data sources to SimaPro 7.3.3 (PRé Consultants 2011). Thus, it is possible to obtain results for other impact categories, but this analysis is out of scope of the project.

3. Results

The animal feed costs ($/kg dry feed) and environmental footprint results of all 180 feeds are shown in Figure 3, 4, 5, 6. The costs of animal feeds vary greatly by region. The prices in the Figure 3 represent 2013 US national averages (Popp 2014). The units for environmental footprint are CO_2e, m^3, and m^2a per 1 kg dry feed. To avoid over cluttering and to enable highlighting of different feed ingredients, we have separated feeds into 3 columns. The first and second columns contain feeds that are the most often used in US swine production; the third column shows results for feeds that are rarely used. This selection was based on discussions with swine producers and nutritionists in Arkansas, so it should not be taken to be representative for the whole US. In addition, the color scheme links up ingredients to specific feed category (e.g., corn and corn by-products).

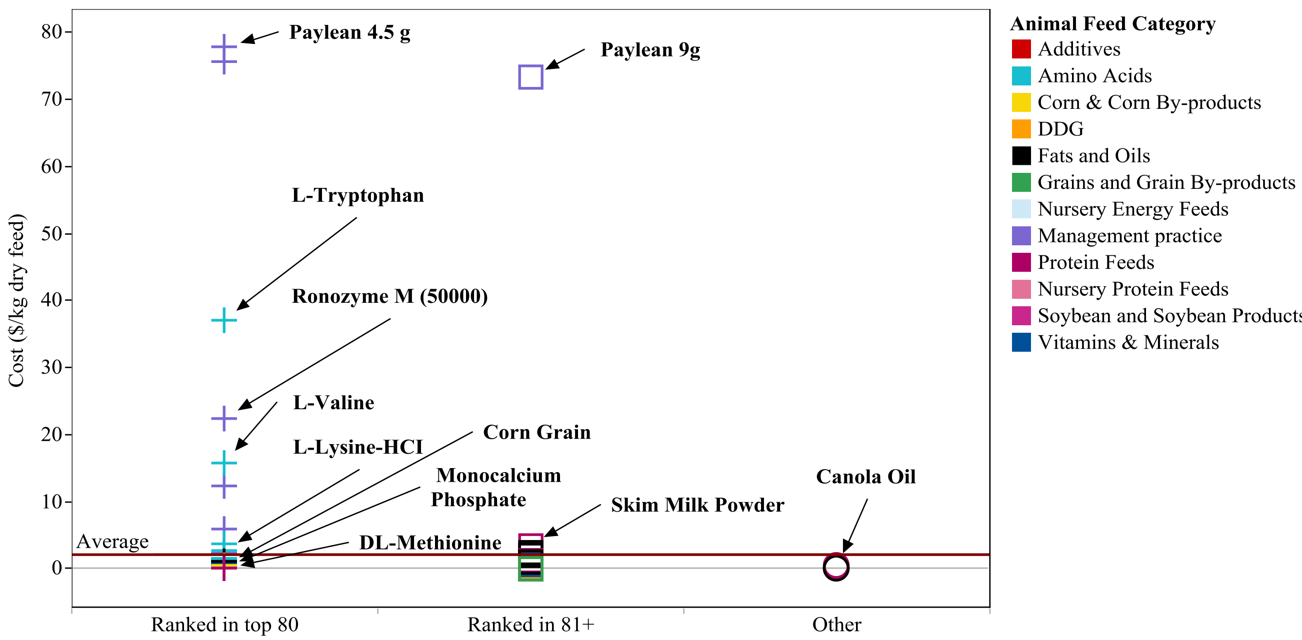

Figure 3. Animal feed cost ($/kg dry feed).

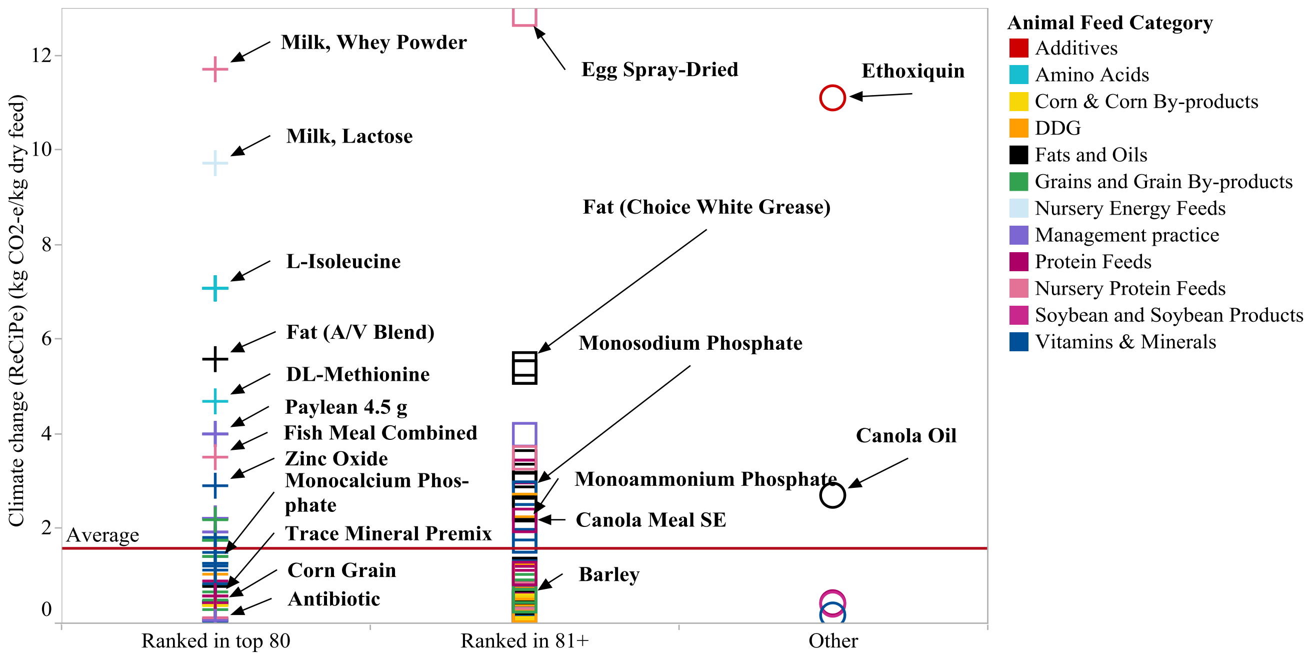

Figure 4. Animal feed climate change impact (kg CO_2e/kg dry feed).

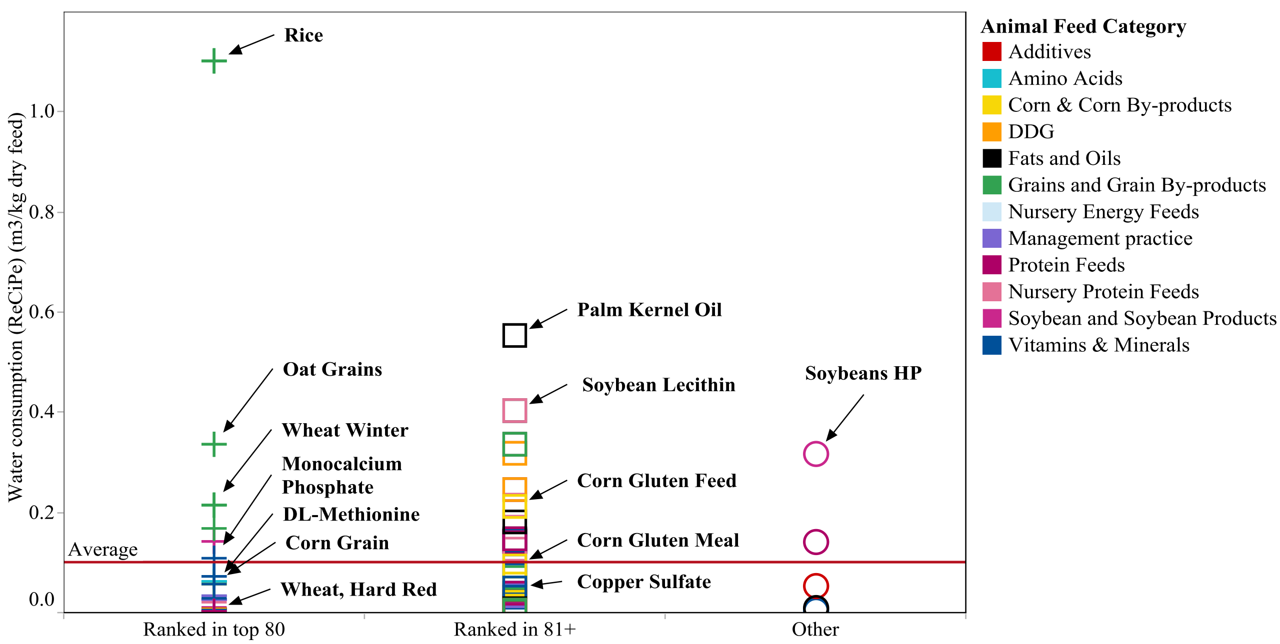

Figure 5. Animal feed water consumption (m^3/kg dry feed).

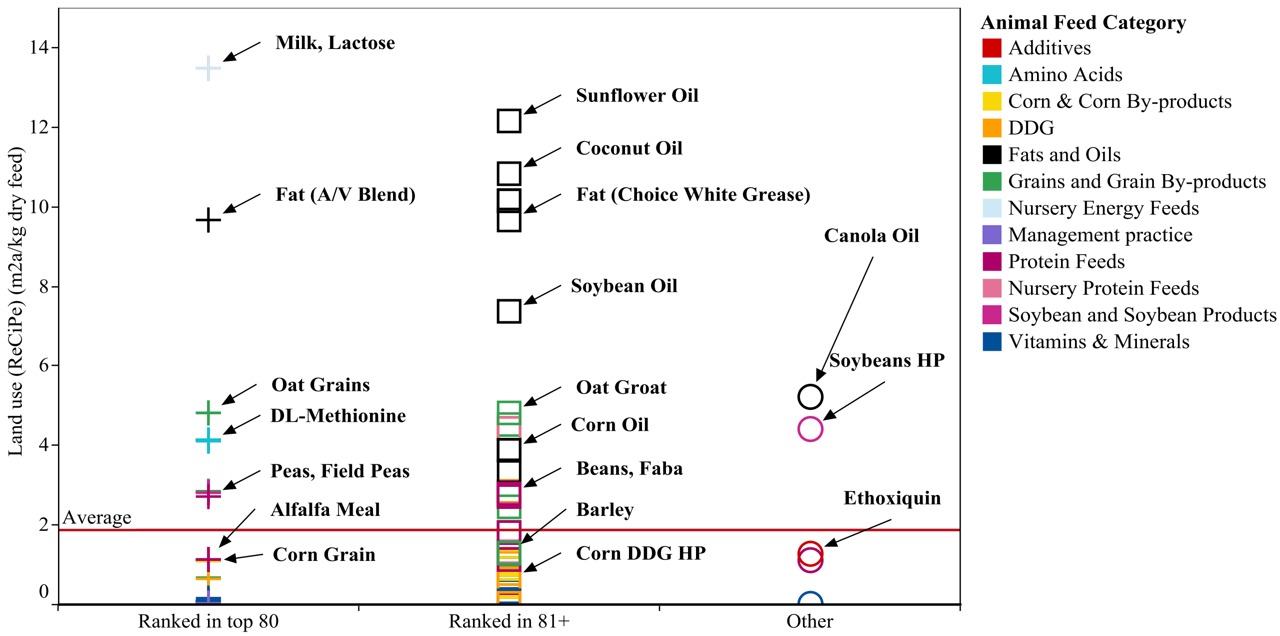

Figure 6. Animal feed land use (m^2a/kg dry feed).

The environmental footprint and feed costs were analyzed against the 7 phase swine diet (nursery-finish), as seen in Figure 7. The reference unit for each phase was defined as \$, kg CO_2-e, m^3, and m^2a per 1 kg of dry feed mix. Formulating swine diets depends on many factors: including but not limited to nutrient requirements, price, feeding limitations, and availability of feeds on the market. Thus, diet is a critical component in modeling environmental footprint and cost. In collaboration with swine producers and nutrition experts in Arkansas, a seven-phase swine diet (nursery, grow to finish) was proposed to serve as a baseline model in PPEFC (Table 2). This diet was used to estimate the environmental footprint of one kilogram of feed mix against the environmental footprint of 1 kg of dried feed.

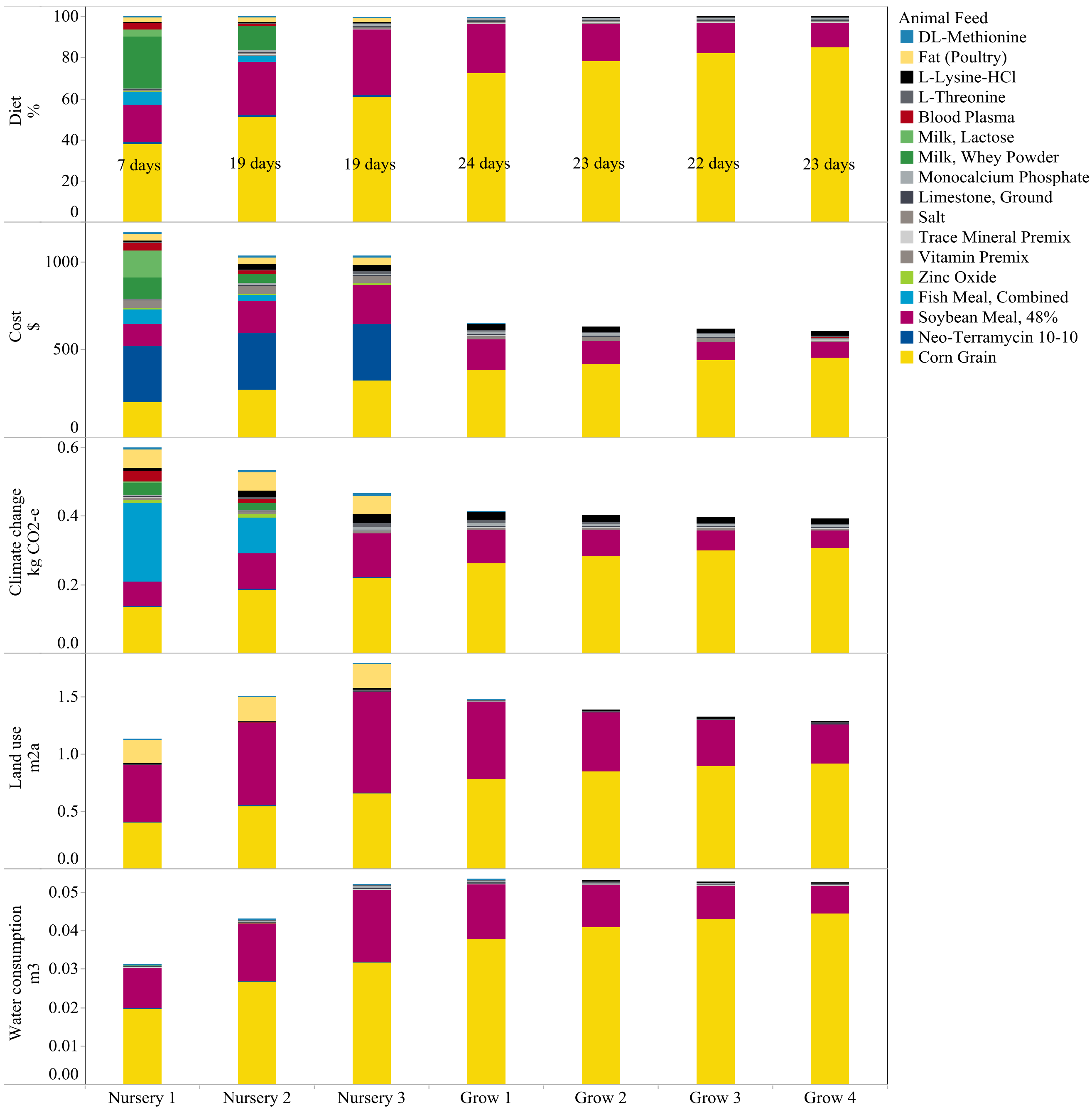

Figure 7. Diet, cost (\$), climate change impact (CO_2e), water consumption (m^3), and land use (m^2a) results of nursery-finish 7 phase swine diet (per 1 kg of dry feed mix).

4. Discussion

Readers should use caution when interpreting results shown in Figures 3, 4, 5, 6 since this is not a direct comparison of values for one feed against another due to different data sources, level of data available, and database limitations. Figures 3, 4, 5, 6 show that the US Animal Feed Database includes a comprehensive collection of animal feed ingredients' environmental footprint, and national average cost results, which will enable farmers using PPEFC to build a variety of swine diets.

There is a distribution of values on an interval for all indicators and one impact category with average values clustered at the bottom range of the interval (e.g., cost). Feeds can have a relatively large contribution to one indicator or impact category and relatively small to another (e.g., sunflower oil). Categories of feeds may show similar results for a specific indicator (amino acids in cost, vitamins and minerals in climate change impact).

Figure 7 shows results for seven-phase swine diet. Large inclusion rates set corn grain and soybean meal as the main contributors for all environmental footprint. Fishmeal use in nursery phase 1 (6.5%) and 2 (3%) is the largest contributor to climate change impact due to higher CO_2-e value per kg of dry fishmeal (note: no water consumption and land use data were available). Inclusion of whey and blood plasma in nursery contributes to climate change impact and for poultry fat to climate change impact and land use inventory. Small inclusion rate of DL-methionine contributes to climate change impact.

5. Conclusion

The importance of the US Animal Feed Database is to compile environmental, economic, and nutrient content in a single location and integrate it into a US LCA - economic model of swine operations. The information from this database will be used as a starting point for identifying potential mitigation options in diet formulation and integration of a linear programming algorithm into the environmental calculator to achieve the necessary nutrient composition under the constraint of least cost and least environmental footprint. During the duration of this project the database will be improved as new information becomes available, such as information gathered from Agri-footprint® (Blonk Consultants 2014). In addition, an explanatory data analysis (EDA) will be performed on the database background results using a graph and network analysis open source tool Gephi (Bastian and Heymann 2009). This analysis should maximize insight into data sets and uncover the underlying structure of the feed database for driving processes and substance. The EDA should help identify outliers and with that, ensure overall quality of the environmental footprint results as well as help to interpret and improve robustness of the results.

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Challenges of comparing food and feed products from different countries of origin

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ABSTRACT

The environmental impacts of home-produced food and feed products as compared to imported products is a widely discussed subject. Within regions such as Central Europe, production systems are comparatively similar. Therefore, inventories representing average production in different countries have to be modeled using detailed agronomic information. In this paper, we describe our approach for compiling life cycle inventories for food and feed products from Switzerland and other European countries. For illustration we present LCA results and their most important drivers. Our results showed that following our approach permits to create inventories for different countries with comparable results, even if data availability is different. The production systems for cereals within Central Europe led to relatively similar LCA results, while the results for potatoes clearly depicted the greater intensity of cultivation in the Netherlands. Transport distances from the production site to Switzerland were decisive only for potatoes.

Keywords: domestic production vs. import, inventories, cereals, potatoes

1. Introduction

The preferability of home-produced compared to imported food and feed products is currently widely discussed in society and politics, especially with regard to the environmental implications. In order to assess the influence of origin on the environmental burdens along the value chain of products, it is necessary to compare agricultural production in different countries. However, within regions such as Central Europe, production systems are comparatively similar. Therefore, for a life cycle assessment of food products, detailed agronomic data from national surveys and other local sources are needed, rather than using rough estimates or data from international sources such as FAOSTAT or FADN, in order to obtain results that depict relevant differences between the countries realistically. However, each country has its own data collection system with its own level of detail. The challenge lies in analyzing those elements that are decisive for differences in the environmental impacts, and in securing the equivalence of systems in a comparative study according to the requirements in ISO (2006). Furthermore, the risk of introducing differences in results as a consequence of different methodologies must be minimized.

We developed an approach for compiling inventories for crops from various European countries processed in Switzerland whose results should be comparable among each other. Those inventories were then used to compare home-produced to imported products via LCA at two levels: 1. At the farm gate: wheat and barley from Switzerland, Germany, and France; potatoes from the same countries and the Netherlands, and 2. at the point of sale in Switzerland: wheat bread, feed barley, and table potatoes. The functional units were 1 kg of product at the farm gate, and 1 kg wheat bread, feed barley, or fresh table potatoes at the point of sale. In this paper, we present our approach to creating comparable inventories of food and feed products using different data sources for each country of origin. We then show LCA results for the products mentioned above, illustrating the feasibility of our approach.

2. Methods

2.1. General approach to create comparable inventories

For all countries of origin, new inventories were created, so as to be able to calculate direct field emissions and the ensuing environmental impacts for all systems with the same tool, ensuring comparability of all inventories. As an example, we compiled our own inventories for crop cultivation in France using raw data from the AGRIBALYSE database (Koch & Salou, 2013) rather than using the AGRIBALYSE inventories themselves. All calculations were done using the SALCA tools (Swiss Agricultural Life Cycle Assessment, Nemecek et al., 2010). SALCA contains a life cycle inventory database for agriculture, models for direct and indirect field emissions, a selection of methods for impact assessment, calculation tools for farm and crop inventories, an evaluation concept, and a communication concept for the results. SALCA requires a set of input data groups to calculate direct and indirect emissions for the inventories: climate and soil properties, yields and information on the crop rotation, fertilization, pesticide use, water use, and field work processes. For every data group, data were available from separate data sources with varying degrees of exactness and specificity in the countries investigated.

In order to secure geographic representativeness as well as comparability for input data, in our approach we suggest a succession of priorities: Priority 1: Data representative for the whole country are to be used if available. Priority 2: If no such data are available, data from the most important production region within the country according to their importance for the total production are to be used. Priority 3: if no country-specific data are available, data from the country with the most similar production system should be used, determining "similar" countries with information from more unspecific data sources such as FAOSTAT for each input data group. Priority 4: Data with little effect on the environmental impacts can also be taken from the SALCA default values, representative for Swiss lowlands, if the effort to collect country-specific data is deemed disproportionately great.

Regarding sources of country-specific data, or data quality, we also followed a set of rules in order to secure comparability between the countries and to ensure, at the same time, representativeness of data for the actual production in the countries: If there were reliable international sources with a quality at the same level or surpassing national data sources, which would cover several of the countries or products under investigation, we used those rather than separate country-specific data. Otherwise we looked for data from national surveys and statistics. If no such data were available, we tried to find other empirical data, or, as the last option, we made assumptions based on national recommendations, e.g. for fertilization.

2.2. System boundaries, allocation, and impact categories

Figure 1 shows the system boundaries for crop production and downstream stages (transport, processing and storage) assumed in this study. Crop production included all processes and inputs occurring on the field plus transport to the farm. Storage and processing was different for each crop: Whereas cereals were first stored at a local place of collection in the countries of origin and then again in Switzerland, potatoes were transported to Switzerland directly. Processing of wheat and barley grains took place in Switzerland. The point of sale for bread and potatoes was a supermarket; feed barley was sold directly at the feed mill.

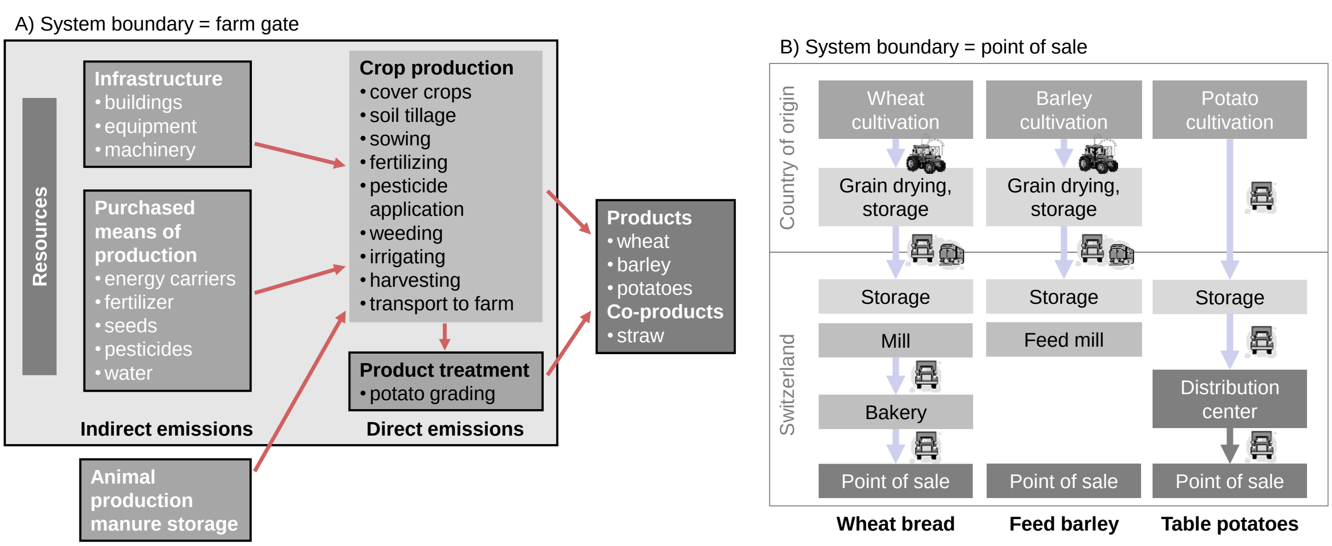

Figure 1. System boundaries for A) crop production and B) food and feed products at the point of sale.

For cereals, the environmental impacts had to be divided between the co-products grain and straw. This was done via economic allocation based on prices (average of three years).

In this paper, we show results for the impact categories energy demand, global warming potential, land competition, water use (weighted with the water stress index, WSI, described in Pfister et al., 2009), aquatic eutrophication N and P, and terrestrial and aquatic ecotoxicity.

2.3. Modelling of crop production

Precipitation and soil data were defined applying priority 2 of our approach: Precipitation data were taken from national climate observation institutions for the major production regions. For clay and humus content, values fitting results from national soil surveys were selected. Other soil data had only minor effects on the emission models; for those we used the SALCA default values which are representative for Swiss lowlands (priority 4). Yields (see table 1) and the share of crop groups in the crop rotation were available from national agricultural statistics, priority 1 being applicable.

Table 1. Crop yields in dt ha^{-1}.

	Switzerland[1)]		**Germany[2)]**	**France[3)]**	**Netherlands[4)]**
	non-extenso	**extenso**			
Winter wheat	66.0	54.7	74.9	71.0	
Winter barley	70.7	52.7	64.8	67.2	
Potatoes	389.4		413.7	449.2	469.0

Data sources: [1)]Farm accountancy data network (Agroscope), AGIS (Federal office for agriculture); [2)]DESTATIS (2012); [3)]raw data for the AGRIBALYSE database (Koch & Salou, 2013); [4)]CBS Statistics Netherlands (2013).

No statistical data were available quantifying the actual amount of fertilizer used in specific crops. Assumptions about nutrient requirement had to be made according to each country's official fertilizing recommendations in accordance with priority 1. These partly led to considerable differences in the overall nutrient requirements of the crops, mostly because the recommendations take account of soil nutrient content in different ways. Likewise, official statistics on the use of manure were lacking. For Switzerland, we had data on manure use in wheat, barley and potatoes from a group of farms which had been investigated in a previous project (Hersener et al., 2011). For France, we used raw data for the AGRIBALYSE database (Koch & Salou, 2013) which contains inventories representing average wheat, barley and potato cultivation based on statistical data and expert estimates. According to FAO (2013), livestock density and farming structures in Germany and the Netherlands were comparatively similar to France and different to Switzerland. Following priority 3, we estimated organic fertilization in Germany and the Netherlands by extrapolating manure use from the French inventories according to livestock density. Likewise, the amount of straw removed from the wheat and barley fields was assumed to be the same in Germany as in France. Table 2 shows our assumptions for fertilization.

Table 2. Assumptions for nutrient requirement and manure use in Switzerland, Germany, France, and the Netherlands.

		Switzerland[1)]		**Germany[2)]**	**France[3)]**	**Netherlands[4)]**
		non-extenso	**extenso**			
Winter wheat	N requirement (kg ha^{-1})	146	135	169	168	
	P_2O_5 requirement (kg ha^{-1})	68	57	66	35	
	K_2O requirement (kg ha^{-1})	97	80	75	37	
	Liquid manure (cattle, pigs) (m^3 ha^{-1})	11.9	11.9	6.5	2.0	
	Solid manure (cattle, pigs, poultry) (t ha^{-1})	0.9	0.9	1.4	1.1	
Winter barley	N requirement (kg ha^{-1})	117	105	148	137	
	P_2O_5 requirement (kg ha^{-1})	74	55	59	39	
	K_2O requirement (kg ha^{-1})	132	98	79	49	
	Liquid manure (cattle, pigs) (m^3 ha^{-1})	6.9	6.9	1.9	0.6	
	Solid manure (cattle, pigs, poultry) (t ha^{-1})	2.6	2.6	2.2	1.8	
Potatoes	N requirement (kg ha^{-1})	100		145	148	246
	P_2O_5 requirement (kg ha^{-1})	61		58	67	83
	K_2O requirement (kg ha^{-1})	252		248	225	176
	Liquid manure (cattle, pigs) (m^3 ha^{-1})	1.2		0.0	0.0	0.0
	Solid manure (cattle, pigs, poultry) (t ha^{-1})	4.7		6.2	4.9	17.7

Data sources for nutrient requirement: [1)]Flisch et al. (2009); [2)]KTBL (2012); [3)]raw data for the AGRIBALYSE database (Koch & Salou, 2013); [4)]WUR (2012).

For pesticide use, we could use data from national survey programs (priority 1). However, these were collected with very different levels of detail and in different time periods in the countries investigated. Swiss data were collected by Agroscope in 2009-2010 at 70-260 field plots, depending on the crop. Data on pesticide use in Germany originated from 37-825 field plots in the years 2007-2011 and were provided by the Julius Kühn-Institut. For France, we used raw data from the AGRIBALYSE database, while the Dutch statistical office provided us with results on pesticide use from 6'822 potato-growing farms in the Netherlands in 2008. In each case, we removed from the input list active ingredients which were only used on a small fraction of the fields so as to avoid distortions of results caused by rarely used and therefore non-representative pesticides. Likewise, we removed active ingredients which were no longer permitted in 2013, according to each country's register of authorized pesticides. Those removed active ingredients were then included into the inventories as "pesticide, unspecified", using average toxicity characterization factors.

Following priority 1, assumptions for the type and number of field work processes were based on national marginal return calculation resources (Boessinger et al., 2011, KTBL, 2012, WUR, 2012), and from the raw data for the French AGRIBALYSE database. The AGRIBALYSE database also contains inventories of field work processes, composed of machinery and diesel consumption adapted to agricultural structures in France. At the same time, the SALCA database contains field work process inventories representing Swiss structures, i.e. smaller machines and higher diesel consumption per hectare due to smaller fields. We worked with the SALCA inventories for Switzerland, while at the other countries we worked with the AGRIBALYSE inventories (priority 3). Lastly, data on water use came from Pfister et al. (2011), who estimated average water consumption for a multitude of crops and countries. We used the "blue water deficit", which is derived by multiplying the theoretical water requirement of the crops by the percentage of surface under irrigation for each country and crop.

2.4. Modelling of transport, processing, and storage

Table 3 gives an overview of the means of transport and the transport distances we used in our calculations. Grain storage was assumed to be six months, half of which took place in the country of origin, the other half at the mill in Switzerland. We used ecoinvent inventories for storage, changing the electricity mix to cover the two storage locations. Grain milling (bread wheat and feed barley) was modelled by adapting the inventory "barley, at feed mill" from the ecoinvent database with expert estimates on electricity, gas, and water use. Processing in the bakery was modelled using inventories from the Danish LCA Food database (Dalgaard et al., 2004).

Potatoes from all countries were assumed to be stored with cooling, in Switzerland for 125 days. The ecoinvent inventory for seed potato storage was used, adapting electricity use according to expert estimates.

Table 3. Means of transport and transport distances for wheat, barley and potatoes from the investigated countries of origin.

Crop	Transport route	Means of transport	Transport distance (km) for crops produced in			
			Switzerland	Germany	France	Netherlands
Wheat[1)]	Farm – local storage	Tractor and trailer	10	25	25	
	Local storage – mill	Lorry and train[*)]	100	300	250	
	Mill – bakery	Lorry	100	100	100	
	Bakery – point of sale	Lorry	25	25	25	
Barley[1)]	Farm – local storage	Tractor and trailer	10	25	25	
	Local storage – mill	Lorry and train[**)]	60	300	250	
Potatoes[2)]	Farm – storage	Lorry, refrigerated	18	600	600	800
	Storage – distribution center	Lorry, refrigerated	49	49	49	49
	Distribution center – point of sale	Lorry, refrigerated	25	25	25	25

Data sources: [1)]Expert estimates (Fenaco); [2)]Expert estimates (Swisscofel) and own estimates with google maps.
[*)]Switzerland: Lorry 25%, train 75%; Germany: Lorry 10%, train 90%; France: Lorry 100%.
[**)]Switzerland: Lorry 50%, train 50%; Germany: Lorry 25%, train 75%; France: Lorry 100%.

3. Results

In general, the agricultural phase dominated most of the environmental impacts. Key factors in the comparison were therefore mostly to be found in the differences between agricultural production in Switzerland and abroad. In a few cases, however, there were also major differences in the downstream stages, particularly with regard to energy demand and global warming potential.

Regarding wheat cultivation, most of the environmental impacts of Swiss wheat were comparable to those of German wheat, but compared to French wheat it was to be rated less favorably regarding many impact categories. Figure 2 shows results for wheat cultivation. In Switzerland, only water use was considerably lower and hence more favorable than in the other countries. Energy demand and global warming potential were higher due to higher machinery input and lower yields in Switzerland as compared to Germany and France. Yield differences were also decisive for land competition. Dates for fertilizer application and differences in the amount of rainfall in winter accounted for slight differences in aquatic eutrophication through N. French wheat caused lower aquatic eutrophication through P because of the low P fertilizer requirement in France. Terrestrial and aquatic ecotoxicity were largely influenced by pesticide use, with very few active ingredients causing most of the impact.

The assessment of bread at the point of sale was consistent with that of wheat cultivation. Although in most instances processing the wheat to bread failed to change the ranking of the countries, the percentage differences between the countries decreased. Transportation had a slightly adverse effect in imported wheat in terms of energy demand and global warming potential.

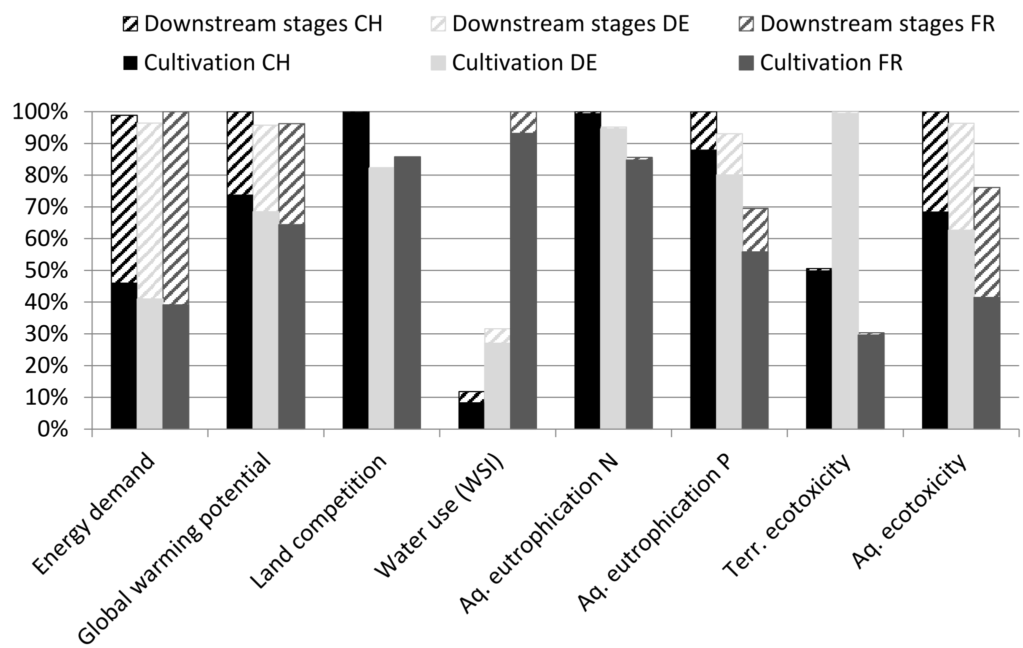

Figure 2. Comparison of the environmental impacts of bread produced from Swiss (CH), German (DE) and French (FR) wheat, at the point of sale in Switzerland. WSI: water stress index.

Results for barley cultivation were comparable to those for wheat cultivation. Nevertheless, Swiss barley was assessed more favorably than Swiss wheat compared to Germany and France, as yield differences between the countries were smaller. Barley from Switzerland required less water than imported barley and had lower ecotoxicity results than French barley (Figure 3). Regarding the remaining impact categories, however, French barley in particular frequently scored more favorably, while German barley was for the most part comparable to Swiss barley. The impact of machinery on energy demand and global warming potential was lower in Germany and France than in Switzerland, which was partly counterbalanced by higher energy demand for irrigation and fertilizer production. Lower P fertilizing requirement in France caused lower aquatic eutrophication P. The use of specific pesticide active ingredients caused differences in ecotoxicity, but fertilizer input, machine usage, and electricity use for irrigation played a role, too. At the point of sale, the country ranking was largely the same as at cultivation level. Shifts in the ranking, for example in energy demand and global warming potential, were caused by different transport distances but were not rated as significant. The share of downstream stages in the environmental impacts of the overall chain was considerably lower for feed barley than for wheat bread, as processing into feed is less complex than bread production.

In potato cultivation, notable differences between Switzerland and Germany were found only in a few environmental impacts (Figure 4). Slightly higher inputs per hectare were offset by higher yields in Germany. Several environmental impacts of cultivation in France were assessed more favorably than in Switzerland, but still both countries were not markedly different for the most part. In France, yields were also higher than in Switzerland, and there was less use of machinery per kilogram of potatoes. Dutch potatoes, in contrast, tended to score worse than those from other countries, particularly regarding nutrient-related environmental impacts. Higher N input, higher organic fertilization, and pesticide use had adverse effects for Dutch potatoes. A better rating could be achieved in aquatic eutrophication P because of the low P fertilization requirement. However, high environmental impacts per hectare were partly offset by higher yields in the Netherlands. At the point of sale the downstream processes, particularly transportation, had a major influence on many environmental impacts. Compared to cereals, this was mainly due to considerably lower environmental impacts per kilogram of produce at the farm gate level. Consequently, regarding the transport dominated impact categories (energy demand, global warming potential, acidification potential), Swiss potatoes rated considerably more advantageously than imported potatoes. In other impact categories, potatoes at the point of sale still followed the rating of the cultivation stage.

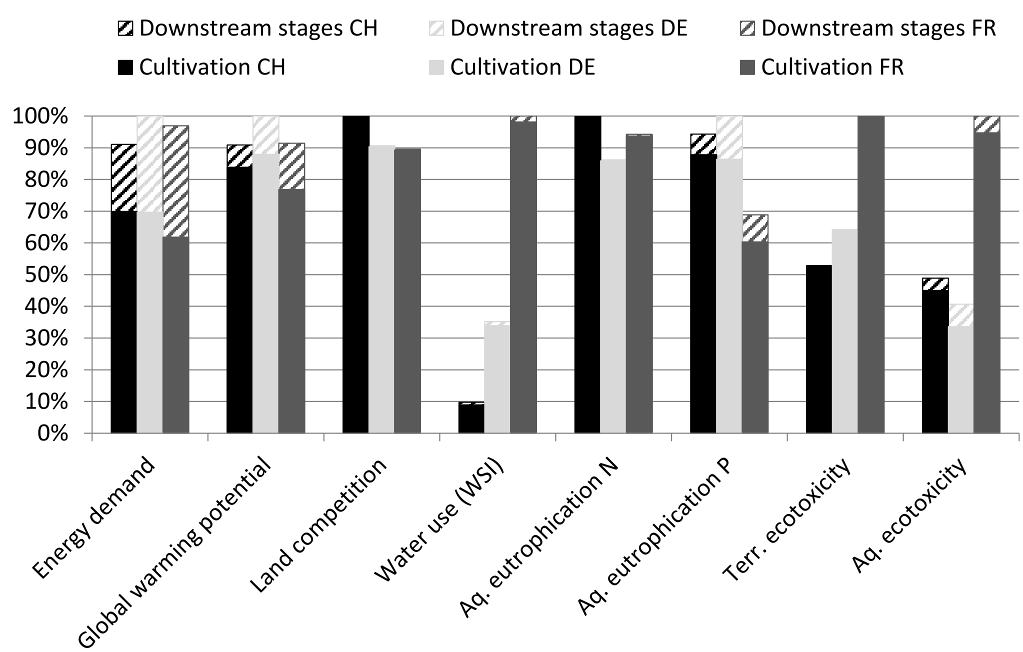

Figure 3. Comparison of the environmental impacts of feed barley from Switzerland (CH), Germany (DE) and France (FR), at the point of sale in Switzerland. WSI: water stress index.

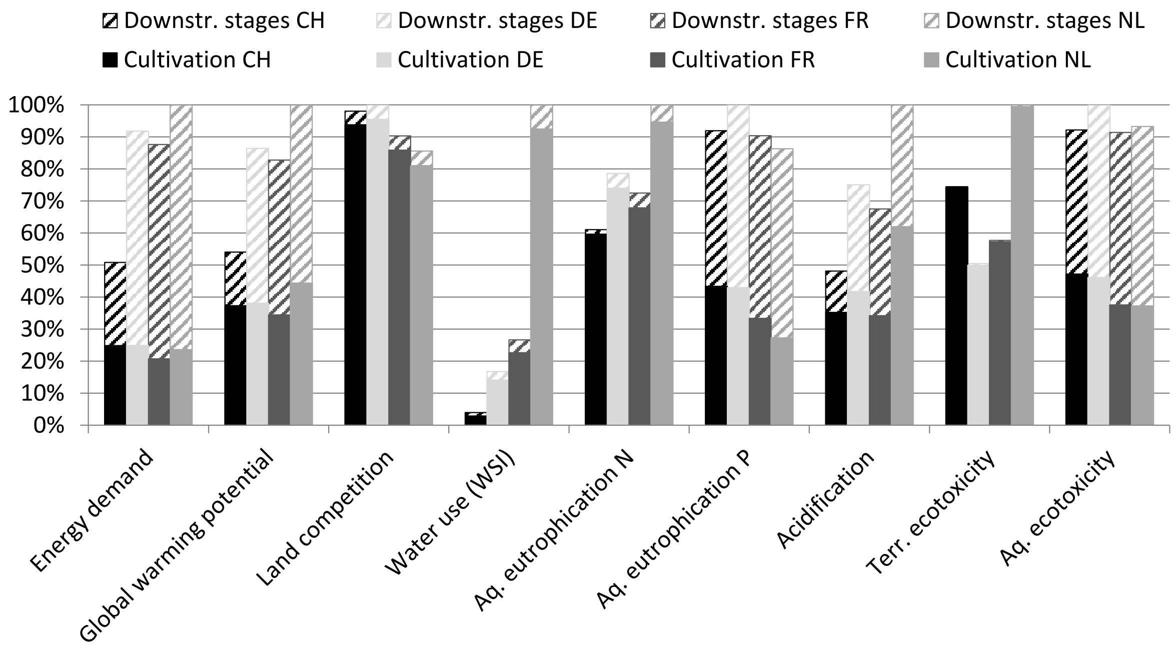

Figure 4. Comparison of the environmental impacts of table potatoes from Switzerland (CH), Germany (DE), France (FR) and the Netherlands (NL), at the point of sale in Switzerland. WSI: water stress index.

4. Discussion

Transport distances can give domestic products a systematic advantage, or at least shift results slightly in their favor, leading to a better rating of home-grown potatoes as compared to imported potatoes. The cultivation stage, however, had a decisive influence on the country rating order in most impact categories, especially for wheat bread and feed barley.

Several major drivers for the evaluation of the cropping stage became apparent. Among these were machine use and fuel consumption. We assumed that in Switzerland fields as well as machines were smaller than in other European countries, which lead to higher impacts of machine use in Swiss production. Indeed, Switzerland ranks among the countries with the highest degrees of mechanization worldwide (Roches et al., 2010), which is clearly reflected by our results.

Water use was the only impact where Swiss production had a systematic advantage compared to the other countries. In addition, irrigation water consumption can have a decisive impact on energy consumption, e.g. for potatoes grown in Switzerland or the Netherlands. The figures we used for irrigation in this study were all taken from the same source, presenting a good basis for a valid comparison between countries. Still they are only estimates that may not always be consistent with water use in reality. Data on irrigation from empirical studies and international statistics (Weber & Schild, 2007, DESTATIS, 2010, FAO, 2013) which were available only for some of the investigated crops and countries point into the same direction that irrigation in Switzerland tends to be lower than in Germany, France, or the Netherlands.

The amount of mineral and organic fertilization affected several environmental impacts significantly. The production of mineral fertilizers and the height of nutrient requirement influenced energy demand, resource use P and K, aquatic eutrophication with P, aquatic ecotoxicity and, partly, human toxicity, while the amount and type of organic fertilizer accounted for differences in terrestrial eutrophication and acidification potential. Using the same type of data source for fertilizer requirement in each country lead to systematic differences, as the national recommendations were based on different assumptions regarding soil nutrient content and requirements per kg crop yield. Thus, P and K requirement was considerably lower in France and in the Netherlands than in Switzerland or Germany. According to EC (2013), phosphorous surplus in soils is particularly high in Switzerland as well as in the Netherlands. If nutrient content of soils would be considered in a uniform manner for all countries, the results of some impact categories could shift. Furthermore, farmers do not always follow fertilization recommendations. In reality the amount of fertilization may differ considerably from the recommendations, leading to different results for the corresponding impact categories. Modelling country averages, the recommendations do already lead to country-specific differences, but investigating data from a farm network would produce more realistic results that could probably vary even more.

For toxicity results, the pesticides applied played a major role. Rather than the overall amount of pesticides, single active ingredients were decisive for the results. Crop-specific data from national surveys were available in all countries, though the scope of those surveys was very divers, leading to great gaps in data quality. We adjusted those differences between the countries by excluding active ingredients which were used only on small proportions of the land or which were not permitted to be used any more in 2013, thus reaching a good level of comparability. Still results might differ from our findings if comprehensive, nationwide surveys were available for all countries.

Lastly, crop yield in the investigated countries was an important parameter for many results. Yield differences between the countries were biggest for potatoes. Negative effects of the input-intensive potato cultivation in the Netherlands were largely offset by higher yields. Yield data were based on resilient data sources which provide great security for this part of the calculations.

5. Conclusion

Our results showed that if our suggested set of priorities for compiling inventories are followed it is possible to create inventories for different countries with comparable results, even if data availability in the countries is quite different. The production systems for cereals within Central Europe led to similar LCA results, while the results for potatoes clearly depicted the greater intensity of cultivation in the Netherlands (higher yields, more organic and more N fertilizer). Furthermore, transport distances were decisive for potatoes. In order to achieve results that are comparable on the one hand and appropriately reflect differences for countries with similar crop production systems on the other hand, it is necessary to treat each group of input data in detail.

With this study we could also point out the importance of high data quality for a systematic and correct comparison between countries. In spite of considerable effort to obtain valid data, we still had to make a lot of assumptions. To increase data quality as well as representativeness of data for the countries, a comprehensive European network of farms would be necessary to collect detailed agronomical data using the same methodology.

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A systems-LCA approach to modelling the impact of improvements in cattle health on greenhouse gas emissions

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ABSTRACT

Poor cattle health reduces productivity, increases mortalities and reduces welfare. These impacts have economic costs, but also increase environmental impacts per unit output. We report the first application of systems-based life cycle assessment (LCA) to modelling the impacts of ten endemic conditions in the UK and their control on GHGE per unit output.

The worst within-herd impacts for milk and beef were Salmonella, bovine viral diarrhoea (BVD), and Johne's disease, increasing GHGE by up to 25% above the healthy baseline. For all conditions, the reductions in GHGE enabled by intervention leading towards recovery were greater than the additional GHGE burdens of implementation. The greatest within-herd reductions were for Johne's, Salmonella and BVD, which may show reductions of up to 20%, while mastitis and lameness are more difficult to treat effectively and thus reduce emissions. Sensitivity analysis showed that effectiveness of interventions was a significant factor in GHGE reductions.

Keywords: cattle health; greenhouse gas emissions; welfare, endemic diseases, systems-modelling

1. Introduction

Poor cattle health reduces productivity, increases mortalities and reduces welfare. These impacts have both economic costs and thus incentives for improvement (Bennett, 2003). However, the environmental impacts of these losses have received little consideration (Stott et al., 2010), although there are expectations for reducing greenhouse gas emissions (GHGE) from cattle production (e.g. GHG Action Plan for England). Reduced productivity can adversely affect greenhouse gas (GHG) emissions (GHGE) per unit output. Veterinary or managerial interventions to mitigate the ill-effects of poor health may themselves cause additional GHGE, e.g. production and delivery of medicine or civil engineering methods to improve welfare. This paper addresses the impacts of ten endemic cattle diseases or conditions on GHGE per unit output, as well as the net benefits of interventions to reduce the impacts of these conditions, using a life cycle assessment perspective and drawing on both statistical and expert veterinary experience. It was part of a wider project, in which the economics of abating GHGE through improved cattle health were evaluated, aiming to understand whether emissions can be reduced in the UK national cattle sector in a cost-effective way by implementing measures to control endemic diseases or conditions.

2. Methods

Expert surveillance and practice veterinarians used statistical and published data, and expert judgement to quantify impacts on productivity, morbidity and mortality. These were translated into parameters to enhance the Cranfield Agricultural Systems-LCA model (Williams et al., 2006). The systems-based nature of the model enabled these impacts to be applied as individual parameters, including increases in mortality rates, reductions in milk yield, daily liveweight gain (DLWG), and fecundity, and increases in feed requirements. A baseline case for healthy cattle was also established, against which the impacts of the conditions could be quantified. This still allowed for mortalities from accidents or other factors, e.g. during calving as well as normal biological variability. These were developed with much veterinary input and the main features were resolved, as shown in Table 1. These represent performance that is above that of the average recorded activity data for UK cattle, as these would be expected to be affected by the ten endemic conditions as well as others outwith this study, e.g. ketosis or tuberculosis. The structure of the LCA model is such that animal production is split by system and age of animal, thus enabling the impacts of diseases to be applied to individual parts of a system or age group. For example, mastitis impacts can be applied purely to the dairy herd, while replacement heifers or beef calves are unaffected and can be modelled as such.

Table 1**.** Main features of cows in the healthy cattle scenario

System	Feature	Values
Dairy	Mortality	1%
	Number of lactations	6
	Lactation yield,	7875
Beef	Mortality	1%
	Number of lactations	8

A set of interventions or mitigation measures was compiled by the veterinarians and the effectiveness of each measure in treating a disease was estimated. These interventions ranged from veterinary, e.g. vaccination or antibiotics; management e.g. better oestrus detection or biosecurity; to engineering, e.g. limestone cow tracks, improved housing ventilation, or pasture drainage. These were applied in the LCA model to calculate the GHGE associated with implementing each intervention. For example, estimates were made as to the number of veterinary visits and treatments that may be required for a given condition, and quantities of raw materials such as ground limestone or sand bedding to improve conditions such as lameness and mastitis were derived. The effectiveness of each measure in treating a disease was estimated to calculate the extent of potential recovery for an infected animal or herd to be returned to the healthy baseline case.

The conditions have different impacts, which thus require different approaches. Reduced growth rate reduces intake commensurately and more time is needed to reach the same end point, so increasing the proportion of energy used for maintenance. Reduced milk yield is more complex. While reduced (e.g. through lameness), energy demand is reduced, but after recovery, the overall lactation yield also depends on management choices. Increased mortality rates were carefully assessed and the model enhanced to allow for more time-critical specification, e.g. higher calf mortality rates increase the number of calf births needed for herd replacements. These tend to occur at a lower age, rather than randomly. So fewer resources are wasted and lower GHGE are incurred than if a random age at death is assumed. Reduced fertility was represented by increasing the calving interval (and increasing AI servicing in dairy cattle). This was based on a stochastic analysis of conception rates. Fighting infection through mounting an immune response demands more energy and effectively increases the maintenance demand for metabolisable energy (ME). There is a lack of evidence on the quantification of ME needs for fighting infection. Conservative estimates were thus made of the expected current levels in commercial herds and how much could be expected for each condition. It must be noted that these ME needs can be high over a relatively short time scale, but are very unlikely to affect, for example, a beef suckler cow for all her life.

3. Results

The GHGE from healthy dairy cows were 0.89 kg CO_2e per kg energy and fat corrected milk. The current national herd performance is 6% higher at 0.94 kg CO_2e. The overall results (ranked by impacts of each single condition) cause increases in the GHGE per unit milk of up to 24%, i.e. for Johne's disease (Figure 1). The maximum impacts for Salmonella, BVD and infertility were also high at 16 to 20%. Liver fluke, IBR, Lameness and Mastitis have more moderate impacts in the range 7 to 10%. Calf diarrhea and pneumonia only affect herd replacements and so have a relatively small impact on GHGE from milk production. These values are the maximum impact for a herd and do not account for prevalence rates, which would reduce the impact nationally. They do, however, indicate clearly how herds can be adversely affected by these conditions and highlights the need for interventions for both welfare and environmental reasons. Of the three highest impacting conditions, infertility is probably most affected by normal management factors, e.g. identifying oestrus in early lactation.

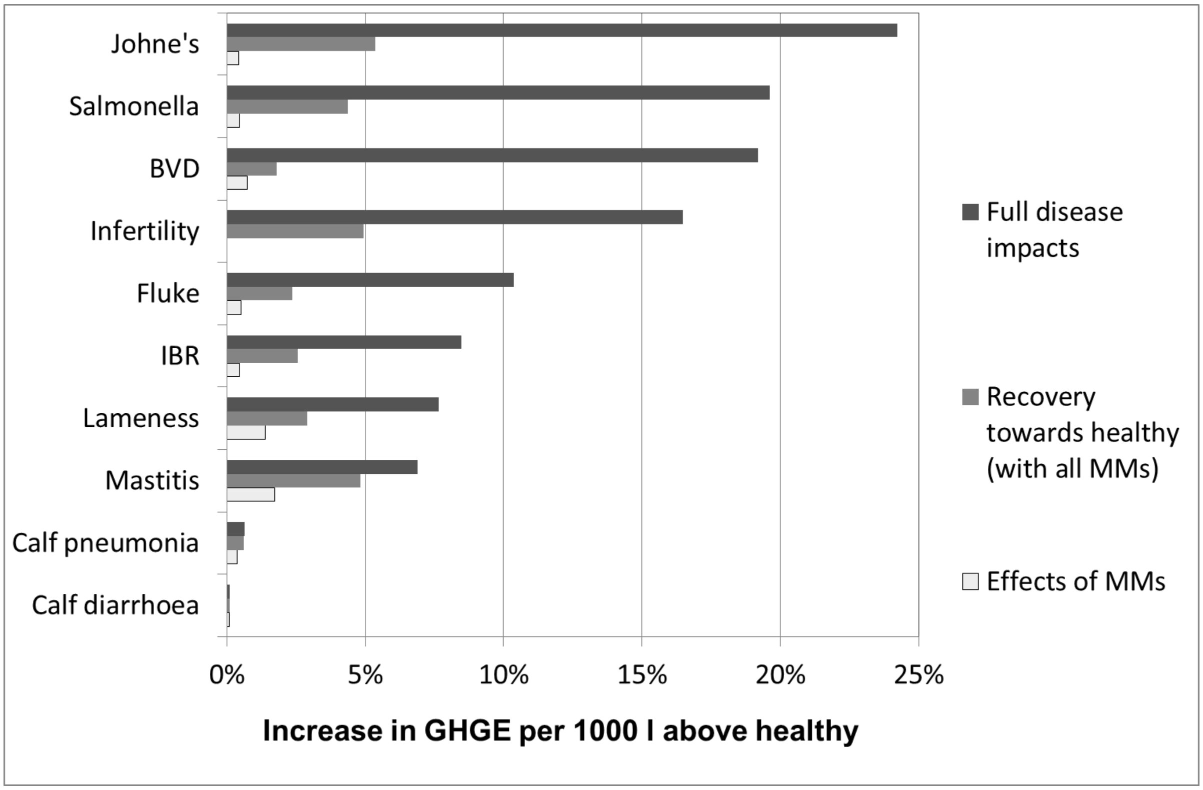

Figure 1. Results for dairy showing percentage increase in GHGE per 1000 l milk above a healthy baseline.

For suckler beef, the GHGE for 1000 kg beef carcass from a healthy herd was estimated to be 17.1 t CO_2e. The current national herd performance is 6.6% higher at 18.2 t CO_2e. The overall results (ranked by impacts of each single condition) cause increases in the GHGE per unit beef carcass of up to 113%, for BVD (Figure 2). This is followed by Johne’s (at 40%), Salmonella, infertility and IBR (at 20% above healthy). Again, these are the maximum farm-level impacts and do not include between-herd prevalence. As with milk, the combined interventions obtained substantial benefits for most conditions. The effectiveness was varied and, for most, reduced the impacts by about 70% of the increase above healthy. Dairy beef results were significantly lower as the breeding phase is accounted for in milk production. BVD remained the worst condition, increasing GHGE by 14% of the healthy baseline for dairy beef production.

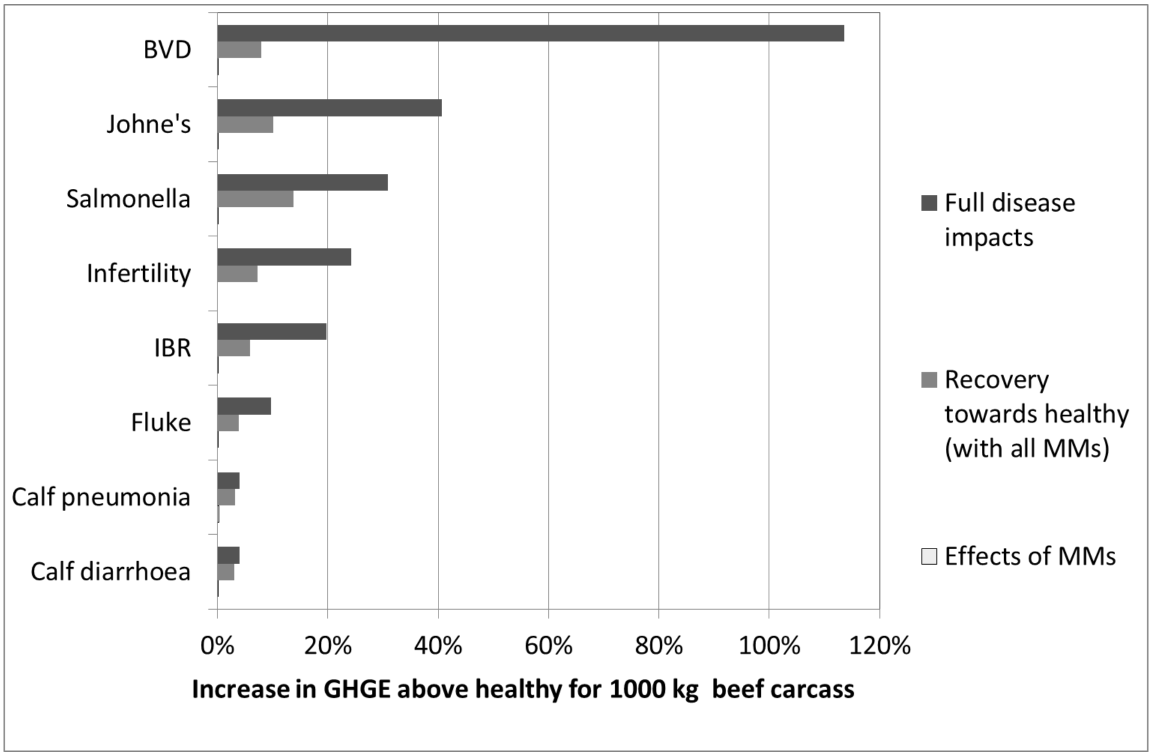

Figure 2: Results of the maximum effects of conditions on GHG emissions from suckler beef production together with the maximum recovery from interventions and the impacts of those interventions themselves.

4. Discussion

The combined interventions clearly produced substantial benefits for top eight conditions and reduced the increase in GHGE caused by the conditions in the range 2% to 5%. Apart from the two low-impacting calf conditions, mastitis is the most difficult to treat effectively and BVD is the least. This does not address economic cost: only the technical feasibility of veterinary, managerial and engineering interventions.

The sensitivity of individual disease impacts on GHG emissions per functional unit were tested by varying the value of one selected parameter in turn by ±10%. For dairy, GHGE per unit milk increased by varying amounts from 0.02% (cow mortality) to 6.2% (extending the calving interval). For beef the effect of increasing metabolisable energy requirement (MER) for both suckler cows and calves have the most substantial effects on GHGE.

A sub-set of mitigation measures were investigated for sensitivity of response. The results ranged from a 0.1% increase in GHGE for veterinary visits to 0.45% for more daily sand use in cow cubicles to reduce the incidence of mastitis. Increasing building replacement rate fell between these measures at 0.13%. Although the environmental investment in a new building is large *per se*, the lifespan is relatively long, at an assumed 25 years, so that the impacts per animal are relatively small.

The systems-based LCA model relies on the disease impact data and expert judgement on the extent to which treatments temper this. In order to provide these estimates, the concept of a healthy animal was used as a reference point and the veterinary experts in the team populated the parameters for each state – healthy, diseased and treated. These data are necessarily informed estimates and do not reflect the considerable variability that would be seen in each of these states on farms. The ambition was to provide a reasonable central estimate which can be modelled to provide high-level analysis of GHGE between conditions.

While previous research has indicated that lameness and mastitis are among the most economically significant endemic cattle diseases, notably in the UK dairy herd, limited GHGE abatement is offered from the controls considered in this study. This owes much to the modest change in productivity parameters which drive GHGE, notably mortality and yield; there is a reasonable degree of additional uptake of controls for these conditions (around 10%).

The key opportunities in terms of GHGE abatement available appears to be with IBR, liver fluke and Johne's disease. These rely on levels of uptake of the controls combined with moderate to high disease impacts on GHGE, while BVD, Salmonella, infertility and some mastitis controls offer moderate abatement levels, combined with moderate to high levels of control uptake.

5. Conclusion

The use of systems-based LCA allowed the effect of individual disease impacts to be quantified and presents scope for application to further diseases (and species). There is further scope to extend the model to consider the interactions between conditions and also between intervention, which would require significant veterinary input and investigation. The findings show the added value in improving cattle health in terms of both productivity and reducing GHGE. The potential to improve animal health and welfare is considerable and is a good example of sustainably increasing production.

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Spatial and Temporal Scale of Eco-label for Agricultural products - case study of milk production

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ABSTRACT

Eco-labels serve as a means to narrow the information gap between producers and consumers with regard to the environmental impact of consumer choices. Whether they are successful is open to debate because of credibility issues, one of which is the specificity of the label: is the label specific to the particular item labelled (e.g. a cut of meat from a specific cow, milk from a particular farm) or the class of items (e.g. beef produced in a country in general, milk from a co-op). Most eco-labels do not provide consumers with information about the specific food products they choose because tactical and operational management decision are not captured in the life cycle inventory collection process, and are usually a historical snapshot from a time in the past. This paper reviews the spatial and temporal aspects of current eco-labels, and assesses the potential to decrease the producer-consumer information gap as farm ICT is developed as part of routing farm management.

Keywords: Eco-label, credibility, ICT, LCA

1. Introduction

In recent decades consumers have started to demand more information about the environmental impact of food products, especially in developed countries represented by organisations such as OECD. Consumers are increasingly willing to buy “environmentally friendly” food products, even if they are more expensive than the market norm (Brécard et al. 2009). At the same time, national policies are driving producers to reduce environmental impacts. Together these have driven demand for informative labelling. Early studies on “environmental labelling” found a wide range of labels with very different meanings and validity (OECD 1991). In 1993, the General Agreement on Tariffs and Trade (GATT) Secretariat differentiated between ‘environmental labelling’ that reflected any type of claim and ‘eco-labelling’ that used life cycle methods (GATT. 1993). Eco-labels are usually voluntary and convey information to consumers about the full life cycle environmental implications of their choices, but the specificity in terms of impact, product and geography can be unclear.

Life Cycle Assessment (LCA) is a tool to assess the potential environmental impacts and resources used throughout a product lifecycle, i.e., from raw material acquisition, via production and use phases, to waste management (ISO14040:2006). As a comprehensive assessment, LCA considers all attributes or aspects of natural environment, human health, and resources (ISO14040:2006), and avoids problem-shifting. There are an increasing number of eco-labels using life cycle methods, though many of them do not follow the LCA standards (ISO 14040/14044). For these eco-labels, the spatial and temporal scale over which they apply is not clear, as is the case for many LCA studies. This can be important in terms of both the impacts and the activity data.

Emerging EU research on deployment of Information and Communication technology (ICT) for agriculture is investigating whether massed sensor technology combined with cloud data sources (sometimes known as the “Internet of Things”, IoT) can provide real-time data for decision making to optimize profit, environmental impact and welfare (for example the ERA-net, Agri-ICT project ‘Sustainable Integrated Livestock Farming’, SILF). SILF is evaluating the concept of using data from sensors with a high spatial and temporal resolution, combined with national databases and life cycle methods to define algorithms for real-time decision support based on the environmental impact of the decision. As such technology develops from concept to application, the question arises whether eco-labelling convey appropriate spatial and temporal information about the specific product being labelled.

This study investigated the how spatial and temporal scale is considered by current eco-labels and evaluated whether integrating the ICT systems and life cycle assessment at farm level may have implications for future eco-labels and the information they convey.

2. Methods

The investigation used three evaluations: (i) review of eco-label types and assessment of scale specificity at the macro level, (ii) evaluation of some example labels commonly used in Europe, and (iii) evaluation of an example farm system, low-cost, grass-based, rotational milk production, for scale specificity and the potential of IoT to permit more specific label information.

2.1. Review of label types

The current status of eco-labels was reviewed. There are three types of eco-labels defined: Type I, Type II and Type III. A comparison with respect to criteria, metrics, standards, verification processes, and spatial and temporal considerations was undertaken. Key issues were identified.

2.2. Evaluation of example eco-labels

Three Type I eco-labels used for food and agricultural products in OECD countries and the Type III international EPD system were evaluated. The selected labels claim to contribute to improved agricultural sustainability. The labels used were:

a) the ***Carbon Reduction label*** was developed by the Carbon Trust, based on the PAS2050 standard and Footprint Expert(TM). The certification uses life cycle assessment including production, use and disposal. The certification must be undertaken every two years and prove that real reductions have been made.
b) the ***MK*** (the abbreviation for Milieukeur) label, was developed in 1995 by Stichting Milieukeur (SMK) using a credit system with points to reward some actions and penalize others. To be eligible to use the MK label, a producer is obliged to have a positive score at the end of the season, both for the company and for each product. The label is based on life cycle methods but not LCA as such.
c) the ***KRAV*** label was first used in 1985 based on IFOAM (International Federation of Organic Agriculture Movements) Basic Standards to fulfil EU regulation (EC) No 834/2007. It is based on life cycle thinking, including environmental impacts and energy consumption.
d) an ***EPD*** (Environmental Product Declaration) is a Type III environmental declaration in accordance with ISO 14025. The EPD system is international, compatible with all types of goods and services, third party verified and a flexible source of information. It is LCA based with defined Product Category Rules (PCR) defined for classes of products that can then be used for the LCA modelling of a specific product.

2.3. Farm system example

Low-cost, grass-based, rotational milk production was taken as an example of a farming system in which tactical and operational decision can have significant influence on the environmental impacts that occur. The system diagram from (Casey & Holden 2005) was evaluated with respect to spatial and temporal specificity.

3. Results

3.1. Label types

Type I labels refer to the environmental quality of a product compared with similar products (or production process) and are meant to encourage a switch towards more environmentally friendly consumption habits. These labels are the result of third party certification programs (usually government supported), and are voluntary. The guiding principles for eco-labels follow ISO 14024, which has more specific requirements than ISO 14020 (ISO14024:1999). Type I labels need to include all stage of the life cycle from resource through extraction, production, distribution and use phase to disposal. Spatial and temporal requirements are not specified in detail, but consideration of relevant local, regional, and global environmental issues when establishing defining criteria and revision period must be specified. ISO 14024 indicates that the ranges and variability of the data obtained for specific products should be analysed to ensure that the selected product environmental criteria are adequate and reflect the differences among and between products. This means that a minimum spatial and temporal resolution should be specified for data quality. Common examples of Type I eco-labels include Nordic Swan and Blue Angle in Europe, but these do not include a specific agriculture or food product category.

Type II labels are self-declarations made by the manufacturers, importers or distributors and refer to specific attributes of the products (like "CFC free", recyclability, degradability and so on). Since these labels are without third party verification, some assurance of reliability is essential. In addition to the requirements of ISO14021, the Type II labels should also follow the principles set out in ISO 14020. ISO 14021 requires Type II labels to take into consideration all aspects of the product life cycle in order to identify the potential for problem shifting. This does not necessarily mean that a life cycle assessment must be undertaken. There are no clear guidelines with respect to spatial and temporal other than stating that the area should correspond to the expected environmental impact and that for comparative claims the calculation should cover an appropriate period such as 12 months (ISO14021:1999). Common examples of Type II labels include Energy Star and WaterSense.

Type III labels are voluntary programs that provide quantified environmental data about a product, under preset categories of parameters set by a qualified third party, based on life cycle assessment, and verified by users or a qualified third party. A typical example is Environmental Product Declarations (EPD). Type III labels are primarily intended for use in business-to-business communication, but under certain conditions, they can be used for business-to-consumer communication (ISO14025:2006). They specifically use the ISO 14040 series of standards either following a complete LCA including goal and scope, inventory analysis (LCI); impact assessment (LCIA) and interpretation or incomplete LCA restricted to goal and scope, LCI and interpretation. There is no clear requirement for spatial and temporal scale to be addressed except to clarify the scope and to define limits, for example, to a certain geographical area. Since Type III labels are based on LCA, the requirement for spatial and temporal specificity should be applied as in ISO 14040/14044. Any lack of spatial and temporal specificity in the LCI introduces uncertainty in the LCIA results. For example, in a milk production system, changes to animal management or fertiliser and slurry application timing can lead to significant change in impact (as discussed later). Currently, the number of Type III label in market is still small. Use in Europe is increasing through the EPD system.

At present, Eco-labels are poorly specified in terms of spatial and temporal detail (Table 1), but recent ICT advances leading to the concept of the IoT might change this. Traceability and sell-by information for specific packaged items are offered to consumers as reliable information so it should also be technically possible for more specific environmental information to be conveyed on labelling.

Table 1. Comparison of the labels types

	Type I	Type II	Type III
Criteria metrics	Multiple	Single	Multiple
Extent of life cycle	Life cycle considered but need not be complete	Life cycle considered but need not be complete	Life cycle assessment, usually complete
Third party verification	Yes	No	Yes
ISO Standard	ISO14024	ISO14021	ISO14025
Spatial aspects	General product characteristics, not specific to the labelled package	Area of environmental impact not specific to the labelled package	Same requirement in ISO 14044
Temporal aspects	General product characteristics not specific to the labelled package	No clear requirement	Same requirement in ISO 14044

A further consideration is that most eco-labels are designed for a specific region or country, which means for example that eco-labels for the China market can be quite different to eco-labels for Europe (UNOPS 2009). (Cohen & Vandenbergh 2012) suggest that the entire life cycle of a product being labelled should be described according to an international standard for measurement and reporting and that once a product meets set criteria, producers can use the eco-label logo for a fixed period of time. Knowledge gained through use of ICT and IoT is increasingly revealing that management and production process change, thus making the label inaccurate. As the technology exists to capture space and time specific data about food products (from precision agriculture technology, activity reporting for regulators and increased ICT use on farms) this weaknesses of eco-labels should be overcome by developing systems to capture real-time data and link it to mass flow during processing.

3.2. Specific examples

The characteristics of the selected eco-labels (Table 2) indicate that all have a relatively loose consideration of spatial and temporal scale that reflects historical technological competence. EPD, which uses ISO 14044 requires specific consideration of spatial and temporal differentiation of the characterization model relating the LCI results to the category indicator and consistency of the spatial and temporal scales of the environmental mechanism and the reference value, but is not specific with regard to the activity data. The Carbon Reduction label that use PAS 2050 requires the time of coverage (age of data and collection period) and geographical specificity (location of collection) to be specified, but this result is then applied to the class of products rather than being product specific.

Table 2. Characteristics of selected Eco-labels

Name of label	Country	Spatial Scale	Temporal Scale	LCA based
KRAV	Sweden	From regional to national	Production phase, historical	LC perspective
Carbon reduction label	International	Product type	Specific to product, historical	LCA
MK	South Africa/EU	National scale, specific to company and product type	Production phase, historical	LC perspective
EPD	International	Same as LCA	Same as LCA, historical	LCA

The life cycle thinking labels (KRAV and MK), do not use LCA but should covers all aspects of production from raw material extraction to production, distribution, use phase and disposal. Spatial and temporal considerations are not clearly stated other than to specify location of production for food categories. MK relies on national data, but for KRAV spatial scale depends on product category. The temporal scale in KRAV and MK only considers the growth stage of products and is historical. Farm ICT and IoT has the potential to improve the information conveyed by the Carbon Reduction label by providing time and location specific data for modelling and labelling. Similar possibilities arise for KRAV and MK. Real time data from sensing, ICT and IoT can be specific to an item within a product class or can be temporally current for classes of product, thus bringing better information to the consumer.

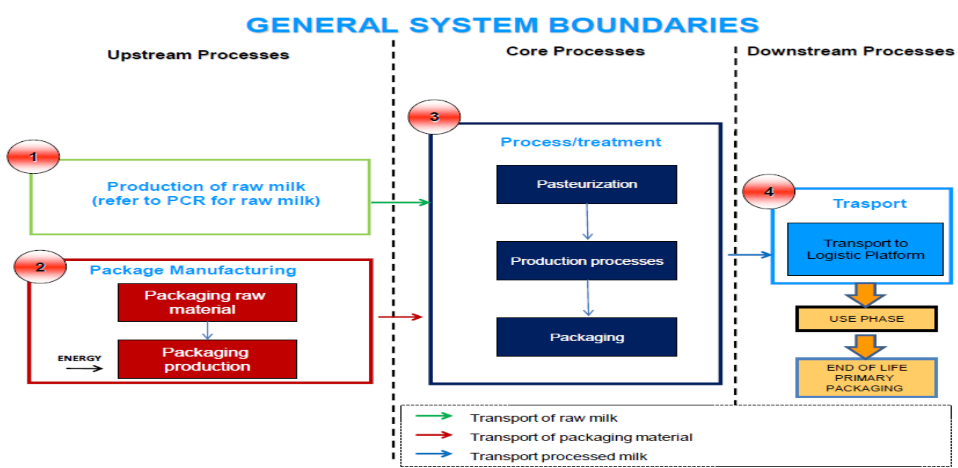

Figure 1. System boundary for Product category rules (PCR) of processed liquid milk and cream

Taking diary products as an example of an EPD, the life cycle has two distinct stages, cradle-to-farm-gate and farm-gate-to-consumer (EPD 2014) (Figure 1). As for many food products, the raw product (in this case milk) production is the main contributor to environmental impacts. The spatial specificity will be limited to the contributing farms, or a sample of the types of farms, while the temporal specificity will be historical, but the sampling period can be of various. A label derived from this system boundary will be spatially specific and historical for packaging and processing, but only the most general information can be included in the downstream processes because a label cannot reflect specific consumer behaviours. It should be made clear however what population the downstream data are derived from as shopping behaviour, transport distances and usage patterns will influence the outcome for this stage.

3.3. Farm system example

The avarage Irish milk production system from “gradle to farm gate” (Figure 2) describes a low-cost, grass-based, rotational dairy system (Casey and Holden, 2005). This system is assumed to be composed of five units: (i) concentrate feed, (ii) fertiliser, (iii) livestock, (iv) manure management and (v) energy. There are different spatial scales using in each part of the system diagram.

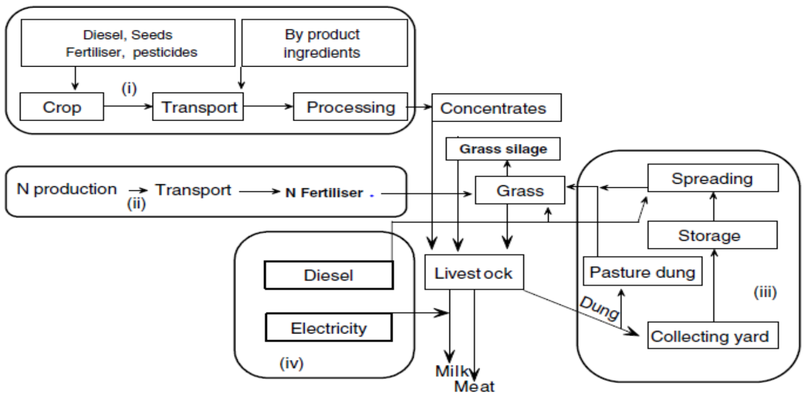

Figure 2. A flowchart of the ‘‘cradle to farm-gate’’ milk production system representing the processes included for describing a typical Irish dairy unit. Where (i) is concentrate feed related, (ii) is fertiliser related, (iii) is manure management related and (iv) relates to electricity and diesel usage (Casey and Holden, 2005).

For concentrate feed, the data will generally be global, using national yield data and management data, and the detail is completely different from that applied in the livestock unit and manure management units. Despite 60% of concentrate feed used in Ireland being imported these data are considered background, yet it is clear that differences in concentrate use will be reposnsible for significant differences between farms. The production of ingredients for concentrated feed accounts for nearly 12% of GWP in this milk production system, and for grass-based systems is likely to be decreased by reducing concentrate feed. Month-by-month, and season-by-season the management of concentrate feed can vary between unprocessed rotations of locally-sourced products to purchased mixes of imported feed. This information cannot currently be conveyed on eco-labels but with modern precision feeding systems it should be possible to collate data at least at farm scale to label milk products. For example, farm ICT can sense the health status of cows for precision feeding to decide the quatity and quantity of supplementary feed. In the context of labels based on EPD (Figure 1), it should be possible to weight contributions from suppliers to dairy processors to link the farm phase to the final product rather than relying on arbitrary historical data.

For the feritiliser unit the impact comes from the decision about which fertiliser to use, when to spread it and how much is used. Fertiliser production accounts for a significant part of the carbon footprint of a farm (14% from prodcution and transportation, 8% from application) therefore tactical and operational management decisions have the potential to siginficantly change the environmental footprint of livestock products. Efficient N management can not only reduce the GWP, but also some local impact categories, like acidification and eutrophication. Modern farm management software and regulatory requirements for N management reporting in Europe (and elsewhere) suggest that more temporally specific data could be associated with products at the farm gate, and then integrated at the processing stage. Modern farm manegement sofatware can use forecasts of polluting losses to support decisions about the best time to carry out slurry application therefore eco-labels should reflect the quality of these decisions in the information they carry.

For the livestock unit, the most important contributor to carbon footprint is CH_4 from enternic fermentation. According to Casey and Holden (2005), a 10% reduction in enteric fermentation from all stock would cause a 5% reduction in kg CO_2 eq $_{ECM}$ (energy collected milk) at the farm gate. It is possible to use known animal attributes collated in livestock registers and known milk production to estimate livestock emissions at the farm scale. As a farmer manages the herd to optimise production this could be reflected in an eco-label because the necessary data are already collected, but are not tagged to the physical product as it goes for processing. At present spatial averages are used on a regional or national scale rather that spatially and temporally specific data.

Farm ICT can identify the breed and age of animals and link to related databases to get the unique animal attributes for estimating the CH_4 emission.

The manure management unit is perhaps less important as its contribution to carbon footprint was estimated at around 7%, thus small changes in management will have relatively little impact, but contribution to acidification and eutrophication is unclear. There is relatively little currently available technology to monitor organic nutrient resources so these are less likely to be considered in the first instance, but as they are the subject of wide ranging regulation this situation might change. It is possible to apply ICT systems on farm for monitoring the emission from manure management (especially storage) and thus real time data collected by ICT could improve the spatial and temporal resolution of the data about manure management phase greatly.

The energy unit includes diesel and electricity consumption. Spatial and temporal specificity could be achieved by using modern farm accounting software to identify consumption from costs, or by using monitoring systems. As this is a relatively small contributor it is less likely to be an immediate focus. The electricity generation for farm can affect its environmental performance in an insignificant way, unless a more energy efficient means of milk preservation is introduced.

4. Discussion

Spatial scale of LCA can be very important for environmental impact assessment. According to (Hauschild 2006), there are three levels of spatial differentiation in LCA, but these tend to relate more to the assumed impact than the detail of the system being modelled.

Site-generic modelling takes all sources considered to contribute to the same generic receiving environment. In this case, we do not have to consider spatial differentiation between sources and subsequent receiving environments. From the analysis of label types, example labels and the example dairy system is it clear that there is the possibility of an over-emphasis on site-generic activity data for food products for pragmatic reasons. While precision may be lost, accuracy may be assured. IoT has the potential to permit eco-labelling systems to move from a reliance on site-generic activity data as it becomes more common in food production systems.

Site-dependent modelling considered some spatial differentiation to distinguish between classes of sources to determine the receiving environment. Source categories are typically defined at the level of countries or regions within countries (scale 50–500 km). The receiving environment is typically defined at high spatial resolution (scale at maximum 150 km, but often down to a few kilometres). With respect to activity data for eco-labelling, some specificity is typical, such as inventory of data for specific suppliers or types of supplier. Again, IoT has the potential to radically enhance the information conveyed on eco-labels so that the data are more focused on the actual suppliers at a specific time.

Site-specific modelling requires large volumes of data to build models for specific locations to evaluate the environmental impacts that are very close to the source. This typically involves local knowledge about the conditions of specific ecosystems that are exposed to the emitted pollutants. This level of spatial differentiation modelling is rare for more than a few processes in a product system. The development of IoT on farms will permit eco-labelling to migrate towards site-specific data and more sophisticated modelling.

Agricultural production systems have different resource consumption (energy, water, natural resource) and emission characteristics depending on the local natural resources, specific management decisions and weather. When compared between different regions or countries eco-labels should reflect these differences, but to date there has been little focus differences between farms and years. A label with site-generic spatial information cannot reflect the true impact of a food product. Farm ICT and sensing has the potential to permit eco-label awarding organizations to demand site-specific data in the near future. Taking dairy farming as an example, the calculation of CH_4 emission from livestock, feed management and fertilizer management should all be possible to ensure products are labelled with information that is specific to the mass flow of product and no more than 12 months old. This improvement of data capture will greatly increase the reliability of information conveyed on eco-labels.

Temporal scale also plays a significant role in the accuracy of LCA, which is generally poor at accounting for time and temporal information (ISO 2006a). The temporal specificity of impact is limited for two reasons. Firstly in the LCI phase aggregated emission data from all the unit processes dispersed through space and time, are typically used (Finnveden et al. 2009, Heijungs & Suh 2002), and secondly during the LCIA phase the potential impacts of the aggregated emissions are assessed using characterization factors based on a fixed time horizon (de

Haes et al. 2002). The temporal specificity of activity data is critical because of internal annual variation in management and changes in technology during the life of the product. For example, the timing of slurry application is very important for GHG emission. N_2O emissions from early time (April) spreading may be 10 times greater than later time (July) spreading (Chadwick et al. 2000), but prolonged storage over the summer period would offset any advantage from late spreading. In addition, the method of application can have a great impact on volatilization losses (Søgaard et al. 2002). When farm ICT can record the time and amount of slurry application the related emissions can be calculated more accurately.

5. Conclusion

This study investigated how spatial and temporal scale is considered by current Eco-label schemes and evaluated whether integrating the ICT systems and life cycle assessment at farm level may have implications for future eco-labels. Considering the extent of application of life cycle assessment to eco-labels for food and agricultural products, it may be premature to talk about consumer attitude towards spatial and temporal scale in eco-labels, but based on the result of this analysis, it will be technically feasibly in the near future to address this matter. If consumers pay more attention to the environmental impacts of their choices they are likely to become more aware of the importance of spatial and temporal scale. The increased popularity of 'local food' suggests that a consumer understanding of scale is now emerging. A label with better-specified spatial and temporal information might be more transparent, thus improving credibility.

Life cycle assessment is a useful tool for assessing the environmental performance of a product or service. In term of accuracy, spatial and temporal scale is important for deriving activity data and for many impact categories in life cycle assessment. It is clear that scale issues are not the focus of labelling, but that technologically it should be possible to forge a closer link between the item labelled and the information on the label. For food products this is particularly important because we know that specific management choices will influence the magnitude of the impacts calculated. Data are increasingly being collected for management, regulation, national statistics and economic reporting, and these will also allow improved calculations for eco-labels if the data are linked to product mass flow. As this is possible for food safety it should be possible for eco-labelling. IoT on farm should in time provide even better information that can be used for eco-labelling.

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Application of Dempster-Shafer theory to integrate methods to propagate variability and epistemic uncertainty in agricultural LCA

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ABSTRACT

We represented variability and epistemic uncertainty in LCA with different approaches in the same framework. Probability distributions are commonly used to represent variability in populations, while fuzzy intervals are an alternative approach for representing epistemic uncertainty in parameters when information is incomplete or imprecise. We used Dempster-Shafer theory to combine both approaches for representing variability and epistemic uncertainty and propagated them with Monte-Carlo simulation through the LCA model. We applied approaches to a case study of dairy farms to estimate their potential direct environmental impacts. Results indicated that consideration of incomplete information greatly increases overall uncertainty in impacts, as measured by a "relative interval width", which was useful for comparing the influence of input uncertainty among impact categories. Thus, our method provides conservative estimates of impacts by considering incomplete information, which is ignored by the classic probability method commonly used in LCA.

Keywords: life cycle assessment, Dempster-Shafer theory, variability, epistemic uncertainty

1. Introduction

Life Cycle Assessment (LCA) is a useful tool to estimate potential environmental impacts and resource use of farming systems (van der Werf and Petit 2002; Thomassen et al. 2008). The reliability of LCA results, which depends primarily on the quality of data and their pertinence for the system studied, is affected by uncertainty (Weidema and Wesnaes 1996; Finnveden 2000). Including uncertainty analysis in LCA may yield results that provide more useful information for decision making (ISO 2006; Heijungs 1996). Therefore, there is a need to improve uncertainty analysis in LCA to increase the reliability of its results.

Uncertainty analysis includes a variety of methods that are used to express and propagate uncertainty in many fields, such as risk analysis (Vose 2008) and LCA (Benetto 2005; Bjorklund 2002). Most studies define two main types of uncertainty (variability and epistemic uncertainty), which have fundamental differences (Morgan and Henrion 1992). Variability (also called stochastic uncertainty) represents inherent differences among individuals in a population. It cannot be reduced but can be represented more precisely if more population data are available (De Rocquigny et al. 2008; Vose 2008). Probability distributions have been used widely in LCA (Basset-Mens et al. 2009; Henriksson et al. 2011; IPCC 2006b) to represent the variability due to randomness in the distribution of a given sample (e.g., with a mean, variance, and normal distribution). In contrast, epistemic uncertainty is defined as lack of knowledge (imprecise and incomplete information) about the true value of a variable or about the system mechanism. It can be decreased if more precise information or more accurate measurement becomes available. In LCA, epistemic uncertainty in parameters is often represented with probability distributions (Lloyd and Ries 2007; Huijbregts 1998), and both types of uncertainty are propagated by Monte-Carlo simulation (MCS), especially in complex models. MCS is an effective and robust way to estimate the uncertainty in predicted potential impacts (Payraudeau et al. 2007; Sonnemann et al. 2003). Some authors (Tan 2008; Reza et al. 2013; Andre and Lopes 2012; Mauris et al. 2001; Chevalier and Téno 1996), however, emphasize the difference between variability and epistemic uncertainty and argue that fuzzy-set theory (Zadeh 1978), with subjective degrees of plausibility/possibility, better represents uncertainty due to imprecise and incomplete information.

Since variability and epistemic uncertainty represent distinct states of knowledge, many studies have modeled them separately in the same framework. For example, Ferson et al. (2002) constructed "probability boxes" by combining probability theory and set theory. Baudrit et al. (2006) represented random variability and imprecision with probability and possibility distributions, respectively, and then propagated them for risk assessment. And Baraldi and Zio (2008) combined MCS and the possibilistic approach to propagate uncertainty. These three studies introduced the Dempster-Shafer theory (Dempster 1966; Shafer 1976) to incorporate imprecise information into probabilistic models, making a bridge that combines uncertainties from different sources (Yager 1987).

Although variability and epistemic uncertainty have been defined, and sometimes analyzed, separately in some LCA studies (Heijungs and Huijbregts 2004; Basset-Mens et al. 2009), few studies (Clavreul et al. 2013) have modeled them in the same framework. The aim of this study is to demonstrate how to combine two types of uncertainty, via Dempster-Shafer theory, to estimate potential environmental impacts of dairy farms. We then compare this method to classic probability methods.

2. Methods

We used probability distributions and fuzzy intervals to represent variability and epistemic uncertainty, respectively. These two types of uncertainty were propagated into LCA results by MCS and interval analysis using R software (R Development Core Team 2012). For each impact category, distributions of impact were represented by a Dempster-Shafer structure and mean impacts were represented by fuzzy intervals. In addition, two other methods for analyzing uncertainty were applied for comparison purposes.

2.1. Representing variability with probability distributions

In a frequentist approach, a probability distribution assigns a probability of any possible event in a random experiment. It is often used to represent the variability of a variable. A random variable X, which is an element of all real numbers ($\mathcal{R}$), has a probability Pr(x) of having value x. In addition, the probability distribution can be described by its cumulative distribution function (CDF) and explained as the probability that X is less than x:

$$F(x) = Pr(X \leq x), for\ all\ x \in \mathcal{R} \qquad \text{Eq. 1}$$

In general, determining a distribution requires empirical data to identify its shape and basic parameters (e.g., mean and variance). If the amount of empirical data is sufficiently large, it can be considered to represent the entire population. However, since data acquisition is often limited by time and cost in LCA studies, probability distributions are generally determined subjectively based on the literature or expert judgment (Heijungs and Frischknecht 2005).

MCS is the most common method for propagating variability to estimate uncertainty in LCA studies. It consists of sampling input variables from their distributions and then calculating potential impacts through the model. By repeating the MCS many times, a CDF can be constructed to predict a probability range that represents overall uncertainty in impacts due to uncertainty in input variables.

2.2. Representing epistemic uncertainty with fuzzy intervals

Fuzzy-set theory is an alternative approach to express epistemic uncertainty. In this approach, an uncertain variable is modeled by a set of “fuzzy” intervals, each with a level of possibility (α) that ranges from 0 (least possible) to 1 (most possible). Denoting each fuzzy interval as $\pi(\alpha_i)$, there is:

$$\pi(\alpha = 1) \subseteq \pi(\alpha_i \in [0,1]) \subseteq \pi(\alpha = 0) \qquad \text{Eq. 2}$$

An uncertain variable can be mapped by a membership function defined by these “fuzzy” intervals and their corresponding levels of possibility. Commonly-used membership functions are shaped as triangles or trapezoids having minimum, maximum and mode values (mode intervals for the latter) (Fig. 1). For example, the minimum-maximum range of variable x, called the “support” (α=0), indicates all possible values of x. The mode (or mode interval), called the “core”, indicates the most likely value(s) (α=1). At any given α level, there is a corresponding interval (called the “α-cut interval”). To propagate uncertainty, the fuzzy intervals of input variables are decomposed at each α level, and interval arithmetic is applied to generate a set of fuzzy intervals of the final result.

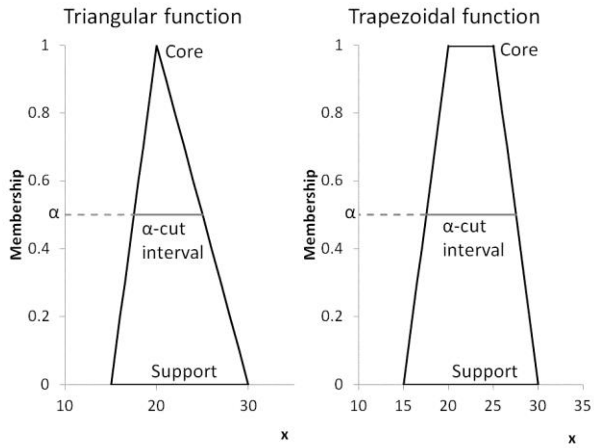

Figure 1. Triangular (left) and trapezoidal (right) fuzzy-interval membership functions for an uncertain variable x defined by core, support and α-cut intervals.

2.3. Dempster-Shafer theory

Dempster-Shafer theory (DST) is a “mathematical theory of evidence” introduced by Dempster (1966) and further developed by Shafer (1976). It is a generalization of discrete probability theory in which probabilities are assigned to sets of values rather than a single value. One important feature of DST is that imprecise information can be used to represent the state of knowledge quantitatively. It includes three basic functions: the basic probability assignment (bpa) function (or “mass function”), the belief function (Bel) and the plausibility function (Pl).

The bpa for a given set *A* (denoted *m(A)*) indicates the proportion of all available evidence that supports the supposition that a particular element of *x* belongs to set *A*. It has axioms such as:

$$m: 2^{\Omega} \rightarrow [0, 1] \qquad \text{Eq. 3}$$

$$m(\emptyset) = 0 \qquad \text{Eq. 4}$$

$$\sum_{A \in 2^{\Omega}} m(A) = 1 \qquad \text{Eq. 5}$$

where 2^{Ω} is the power set that comprises all possible subsets, including the empty set $\emptyset$. *A* is any subset (called a “focal element”) of power set. The belief and plausibility functions are defined from the bpa. The belief function of *A* is the sum of the bpa of all of the subsets (*B*) of *A* ($B \subseteq A$):

$$Bel(A) = \sum_{B \subseteq A} m(B)\,, B \text{ is all of the subsets of } A, \text{and } B \neq \emptyset \qquad \text{Eq. 6}$$

The plausibility function of *A* is the sum of the bpa of any subset (*C*) of power set with the condition that the intersection of *C* and *A is a non-empty set* ($C \cap A \neq \emptyset$):

$$Pl(A) = \sum_{C \cap A \neq \emptyset} m(C), C \in 2^{\Omega}, \text{and } C \neq \emptyset \qquad \text{Eq. 7}$$

The three concepts can also be used in continuous probability distributions where any element of the uncertain parameter *x* is expressed as an interval ($[a_i, b_i]$) with bpa$_i$ (where $a_i \leq b_i$ for all *i*). Thus, the power set *A* is the collection of these intervals with their corresponding bpa$_i$, and the sum of bpa$_i$ equals 1. So, Eq. 6 and 7 can be transformed as:

$$Bel(x \in]-\infty, x]) = \sum_{b_i \leq x} m([a_i, b_i]) \qquad \text{Eq. 8}$$

$$Pl(x \in]-\infty, x]) = \sum_{a_i \leq x} m([a_i, b_i]) \qquad \text{Eq. 9}$$

The belief and plausibility functions can be considered the lower and upper probabilities of a given value of x (Ferson et al. 2002), respectively, and the true probability distribution of x ($Pr(x)$) lies inside them, interpreted as:

$$Bel(x) \leq Pr(x) \leq Pl(x) \qquad \text{Eq. 10}$$

with an interval of the mean of x

$$\sum m_i a_i \leq E(x) \leq \sum m_i b_i \qquad \text{Eq. 11}$$

If uncertain parameter x is determined by a single set of values ($a_i = b_i$ for all i) instead of intervals, the belief and plausibility functions converge on the same distribution, as in a classic CDF.

We considered fuzzy intervals of uncertain parameters as focal elements, used DST to combine variability and epistemic uncertainty in input-parameter values, and propagated them with MCS. Thus, assuming that impact category (y) is calculated by the model ($f(x_1, x_2 \ldots x_k, x_{k+1}, x_{k+2} \ldots x_n)$), we:

1. Randomly generate a matrix (B rows (10,000) × k columns) of the first k variables using each one's probability distribution while preserving correlations between them: $M_b(x_1, x_2 \ldots x_k)$.
2. Select a possibility level α_i (e.g., assign values from 0 to 1 with step 0.1) and its corresponding fuzzy interval ($\pi^{k+1}, \pi^{k+2} \ldots \pi^n$) for the last n-k uncertain parameters ($x_{k+1}, x_{k+2} \ldots x_n$).
3. Using the variables in each row of $M_b(x_1, x_2 \ldots x_k)$, calculate minimum and maximum values of y with all possible combinations of the last n-k variables ($= 2^{(n-k)}$) in α_i, with a lower bound of $L(y) = \max [f(x_1, x_2 \ldots x_n)]$ and an upper bound of $U(y) = \min [f(x_1, x_2 \ldots x_n)]$. For a model with only monotone functions, the optimization algorithm can be simplified by interval analysis.
4. Repeat steps 2 and 3 for all α_i to generate the fuzzy intervals of y_b ($[U(y), L(y)]_b$) and find the support of y_b (denoted $\pi (y_b)_{\alpha=0}$). Then attribute a mass ($m_b = 1/B$) to the support.
5. Repeat steps 2-4 B times to obtain a set of supports ($\pi_1, \pi_2, \ldots, \pi_B$) as focal elements, which is used to construct lower (belief function) and upper (plausibility function) bounds of y using Eq. 8 and 9.
6. Calculate the mean of the lower and upper bounds of y using Eq. 11 for each α to generate fuzzy intervals of the mean of y.

For example, assume that indicator Y is modeled by a function with two uncertain parameters (X_1 and X_2), $Y = X_1 \times X_2$, where X_1 is normally distributed ($X_1 \sim N_{prob}(100,20)$) and X_2 is modeled by a triangular membership function ($X_2 \sim T_{fuzzy}(2, 6, 3)$). Thus, we construct the lower and upper bounds of Y and fuzzy intervals of the mean of Y at each α (Fig. 2) by following the above steps. Consequently, this procedure generates a set of intervals (11, in this study) with their corresponding α, and the mean of Y is represented as a membership function of this fuzzy set that is determined by its support, core and other α-cut intervals.

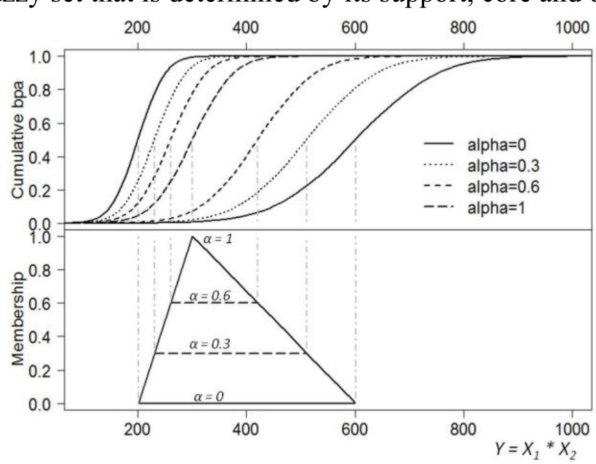

Figure 2. (Top) Lower (belief) and upper (plausibility) bounds of indicator Y at different possibility levels (α) and (bottom) fuzzy intervals of the mean of Y with support (α=0), core (α=1) and two α-cut intervals (at α=0.3 and 0.6).

2.4. Case study

We constructed an LCA model to estimate environmental impacts (climate change, acidification and eutrophication) of on-farm emissions of dairy farms. The functional unit was 1 metric ton of fat-and-protein corrected milk (FPCM). This model was based on the EDEN-E (Evaluation de la Durabilité des ExploitatioNs) tool, developed previously to estimate LCA-based environmental impacts of individual dairy farms (van der Werf et al. 2009). In this study, we focused only on direct impacts of the milk-production subsystem, because they were affected directly by uncertainty in emission factors. We used data from 41 conventional dairy farms from EDEN-E datasets. We obtained input variables such as animal production (e.g., meat, milk), number of animals by age and sex, and usable agricultural area. Other variables such as quantities of nitrogen (N) in farm inputs and outputs (e.g., fertilizers, feed, waste, animals), energy agents (e.g., diesel, gasoline, electricity), lubricants and plastics were also taken from EDEN-E. In addition, emission factors were used to estimate gaseous emissions; their default values and ranges of uncertainty were taken from the literature (IPCC 2006a; EMEP-CORINAIR 2001). These variables were used in the model to estimate direct impacts of conventional dairy farms. We considered two types of uncertainty in input variables: variability in structural characteristics of sampled farms and epistemic uncertainty in emission factors. Impact categories were calculated by multiplying emissions with the characterization factors of the CML-IA database (Guinée et al. 2002).

To compare results with those of the classic probability method, we made three scenarios to analyze uncertainty in the LCA model. For all scenarios, we used truncated normal or uniform distributions to represent variability in each farm characteristic because they provided only non-negative values. Means and standard deviations for each characteristic were determined from the empirical EDEN-E sample. Correlations between these variables were preserved using Spearman rank-order correlation (Helton and Davis, 2003). In the first scenario (S1), uncertainty in emission factors was ignored (default constants used). In the second scenario (S2), we considered uncertainty in emission factors with triangular probability distributions. In S2, default values and ranges of uncertainty were used as the mode and minimum/maximum values in the distribution, respectively. In the third scenario (S3), we used fuzzy intervals with triangular membership functions to represent uncertainty in emission factors. The same default values and ranges of uncertainty were used as the core and support in the membership function, respectively. Indeed, S1 can be considered a special case of S3 that considers only the core interval (α=1). Each emission factor was assumed to be independent. Scenarios S1 and S2 used MCS to propagate uncertainties, while S3 combined MCS and interval arithmetic to generate an interval distribution. Simulations were repeated 10,000 times. In S3, finding minimum and maximum impact values for each of the 10,000 replicates theoretically required calculating all possible combinations of emission factors (2^m combinations, m = number of emission factors), but doing so would have increased calculation time considerably. Therefore, since the LCA model was monotonic (emission factors used only addition and multiplication), calculations were optimized by using the minimum and maximum of each α-cut interval of emission factors.

Statistics (mean, 5th and 95th percentiles) of impact indicators were calculated as single values in S1 and S2 and as intervals in S3. To compare uncertainty in mean impact between categories, we calculated a "relative interval width" (RIW), equal to the maximum of a statistic's interval minus its minimum, divided by its mode. Thus, the RIW of mean impact in S3 was the width of the indicator's mean interval (i.e. its support) divided by its core (i.e. the most likely mean value). Because uncertainty in emission factors was assumed to be zero in S1 and a known distribution in S2, they were considered to have intervals of zero width.

3. Results

Statistics of the three impact categories differed by scenario (Table 1). For all three impact categories, the difference between the 5th and 95th percentiles ($I_{percentile}$) in S1 and S2 was narrower than the difference between the minimum of the 5th-percentile interval and the maximum of the 95th-percentile interval in S3, indicating higher overall uncertainty in S3. The increase in overall uncertainty was due to inclusion and representation of epistemic uncertainty in emission factors. The percentage increase in $I_{percentile}$ from S1 to S2 was 25% for climate change, 80% for acidification, and 0% for eutrophication, while that from S1 to S3 was 212% for climate change, 372% for acidification and 29% for eutrophication. Thus, overall uncertainty in impacts increased greatly when uncertainty in emission factors was considered as imprecise information.

Table 1. Statistics of potential climate change, acidification and eutrophication impacts per t of fat-and-protein-corrected milk (FPCM) in three scenarios that represented uncertainty in emission factors (EFs) differently: S1 - no uncertainty in EFs, S2 - probability distributions for EFs, S3 - fuzzy sets for EFs.

	Climate change (kg CO_2 eq./t FPCM)			**Acidification (kg SO_2 eq./t FPCM)**			**Eutrophication (kg PO_4 eq./t FPCM)**		
Statistics	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**
Lower limit (5th percentile)	775	852	[511, 1383]	8.1	9.5	[3.6, 18.2]	4.3	3.9	[2.7, 5.0]
Mean (support of the mean)	1065	1208	[718, 1874]	10.8	14.3	[4.8, 24.3]	8.6	8.2	[6.6, 9.4]
Upper limit (95th percentile)	1400	1632	[947, 2459]	14.2	20.5	[6.3, 32.4]	13.7	13.3	[11.4, 14.8]
Relative interval width of the mean	0%	0%	89%	0%	0%	134%	0%	0%	35%

When visualized as CDFs (Fig. 3), S1 and S2 were represented by a single CDF each, while S3 was represented by two CDFs (plausibility and belief functions) defining upper and lower bounds of impact in each category (Fig. 3). These bounds were more widely separated for acidification and climate change impact than eutrophication.

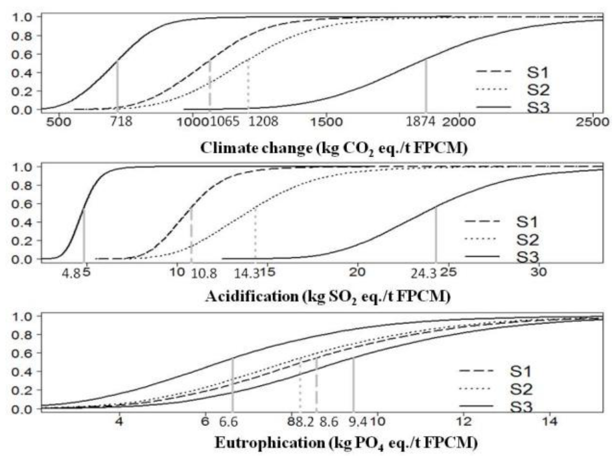

Figure 3. Cumulative density functions of direct (on-farm) climate change, acidification, and eutrophication impacts per t of fat-and-protein corrected milk (FPCM) in three scenarios that represented uncertainty in emission factors (EFs) differently: S1 - no uncertainty in EFs, S2 - probability distributions for EFs, S3 - fuzzy sets for EFs (solid curves bound 90% of possible values). Vertical gray lines indicate the mean impact (support of mean impact in S3) of each scenario.

Membership functions of mean impacts in S3 were nearly triangular, with minor skewness (Fig. 4). For example, the support of climate change ranged from 718-1874 kg CO_2 eq./t FPCM, with a core of 1065 kg CO_2 eq./t FPCM. Note that the core of mean impact in S3 equaled the mean impact in S1, for which default values (considered as the true values) were used for emission factors. RIWs of mean impacts in S3 indicate that uncertainty in mean acidification impact (134%) was larger than that in mean climate change (89%) or eutrophication (35%) impacts (Table 1). However, S1 and S2 each had a single mean (e.g., 1208 and 1065 kg CO_2 eq./t FPCM, respectively) and RIWs of mean impact of 0%.

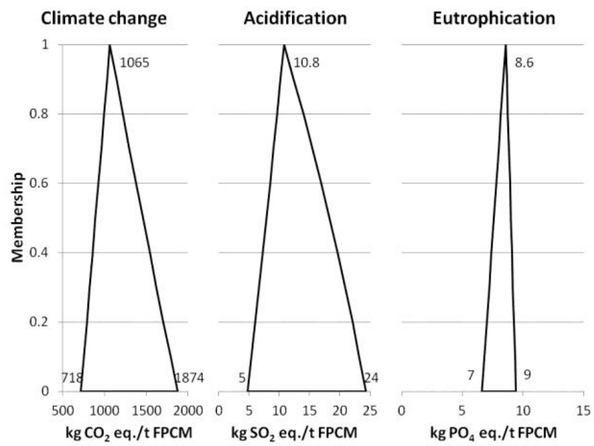

Figure 4. Fuzzy-interval distributions of means of climate change, acidification, and eutrophication impacts per t of fat-and-protein-corrected milk (FPCM) in scenario 3 (based on Dempster-Shafer theory).

4. Discussion

We focused on two sources of uncertainty in this case study: variability (in farm characteristics) and epistemic uncertainty (in emission factors). The classic probabilistic approach expresses both epistemic uncertainty and variability with probability distributions; however, subjectively defining probability distributions may underestimate overall uncertainty. Therefore, unknown distributions should be considered as another source of uncertainty due to incomplete information and modeled with fuzzy intervals. If both probability distributions and fuzzy intervals exist in the same analysis, our DST-based method can combine them to estimate overall uncertainty in impact (S3). It provides a more conservative range of uncertainty (i.e. the interval between the minimum of the 5^{th} percentile and maximum of the 95^{th} percentile) than the classic probabilistic approach. Since S1 considered only variability in farm characteristics, the increase in overall uncertainty in impacts in S2 and S3 compared to S1 reflects the contribution of epistemic uncertainty in emission factors alone. Considering emission factors as fuzzy intervals (S3) increased overall uncertainty more than considering them as random values (S2). Unlike variability, epistemic uncertainty in emission factors can be reduced when more precise information becomes available. For example, any distribution inside the bounds of plausibility and belief yields a mean value and a smaller range of uncertainty, because the true values (S1) or distributions (S2) of input variables are known or assumed to be known.

In parallel, the imprecision in emission factors was propagated into mean impacts, which were constructed from membership functions of their fuzzy sets. Those who used a similar propagation procedure in LCA (Clavreul et al. 2013) or risk assessment (Baudrit et al. 2006) studies considered all fuzzy intervals of impact as a set of random intervals with equal probability and then calculated a weighted-mean interval from upper and lower bounds of distributions. In contrast, we separated this process into two steps: (1) estimate the overall range of all possible impact values using the supports of emission factors and (2) model mean impacts with fuzzy intervals. Indeed, showing the membership function of mean impact instead of a weighted-mean interval provides more information to decision makers, such as the levels of possibility corresponding to the most likely mean impact and mean impacts of best- and worse-case scenarios based on the degree of possibility. This information allows a more precautionary approach than a simple interval of mean impact for evaluating the magnitude of and uncertainty in predicted impacts. For fuzzy intervals, the RIW of mean impact is a comparative indicator that reflects the influence of explicitly representing the knowledge of information as incomplete (an unknown distribution), unlike the classic probability method, which ignores this source of uncertainty. It enables the influence of epistemic uncertainty on different impacts to be compared when calculating a coefficient of variation is difficult or complex (e.g., in S3, which comprised multiple probability distributions). Comparing RIWs among impact categories illustrates the relative influence of epistemic uncertainty on overall uncertainty in the impacts of each. If overall uncertainty is hindering decision making, this information could lead decision makers to focus on reducing the sources of epistemic uncertainty that contribute the most to uncertainty in impacts. For example, since epistemic uncertainty in emission factors had a larger influence on acidification than eutrophication impacts in this study, more precise measurement of acidification-related emissions would have a relatively larger influence in reducing overall uncertainty in an impact.

For the sake of simplicity, we illustrated a simple LCA example based on a previous work. It had far fewer variables and parameters than a full LCA study; in addition, the model was monotonic, which simplified the optimization algorithm in the simulation. Increasing the number of uncertain variables (especially imprecise variables) may increase the complexity of computation and even greatly overestimate uncertainty. Therefore, performing an initial sensitivity analysis of the LCA model is recommended (Heijungs 1996; Henriksson et al. 2013) to focus on the input variables that influence potential impacts the most. This decrease in the number of uncertain input variables may accelerate the optimization algorithm, especially in more complex and non-monotonic models.

Correlations between random variables (i.e., inter-farm variability) were preserved in this study, while independence was assumed between random variables and imprecise parameters (emission factors). This assumption allowed a conservative confidence interval of impacts to be generated for all three scenarios, because all possible combinations of input variables were included in the stochastic simulation. However, representing dependence between random variables and imprecise parameters (if known) could improve the precision of predicted impacts. More research is needed on this issue to improve the validity of LCA results.

The DST-based method constructs two boundary distributions using belief and plausibility functions. This structure has been interpreted as "imprecise probability" (Ferson et al. 2002), which covers all possible probability distributions. Although it reflects the true state of knowledge (e.g., incomplete information about emission factors), an extremely wide range of potential impact is likely to be less useful for decision makers. Thus, simplifying interpretation of results by decision makers remains an open question. To address this problem in LCA, Clavreul et al. (2013) calculated a "confidence index" (Dubois and Guyonnet 2011) to generate a weighted probability distribution. Decision makers can choose this confidence index subjectively, depending on whether their decision policies are more optimistic (close to the upper bound) or pessimistic (close to the lower bound). We concur that this kind of confidence index is useful for decision making in LCA studies when uncertainty is modeled with imprecise and incomplete information.

5. Conclusion

The classic probability method is rigorous in that it requires precise information to express an uncertain variable, but subjective assumption about its distribution may underestimate uncertainty in the predicted result. In addition, it cannot separate epistemic uncertainty from variability, meaning that decision makers will have no information about the relative influence of each on overall uncertainty. Our proposed method overcomes this limit by integrating fuzzy intervals to represent imprecise data (e.g., emission factors) in probability models. As a consequence, a distribution with two bounds and fuzzy intervals of mean impact was generated. Combining the effects of variability and epistemic uncertainty yields a wider range of potential impacts, which may influence decision making. Fuzzy intervals of mean impact model the uncertainty in mean values. The RIW of mean impact, as a comparative indicator, reveals the influence of epistemic uncertainty on uncertainty in impacts, which may help decision makers adopt appropriate strategies if they want to improve the reliability of LCA results.

The paper demonstrated the application of DST in uncertainty analysis in a simple LCA study. Representing the lack of knowledge as fuzzy intervals differs from treating it as randomness. Thus, it provides a conservative but robust way to represent the state of knowledge in LCA studies when information is scarce. Its application in LCA is currently limited, however, due to its greater model complexity and, if epistemic uncertainty is large, greater difficulty in distinguishing potential differences among scenarios. Considering dependence among input variables is also important, since it gives more precise results, but techniques for doing so with fuzzy-interval variables are difficult or complex. Therefore, more research is needed to focus on these issues to improve the feasibility of this method in LCA.

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Comparison of different calculation procedures and emission factors in the manure management systems of swine production

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ABSTRACT

The aim of this paper was to perform a sensitivity analysis of different calculation procedures and emission factors to estimate methane (CH_4) and nitrous oxide (N_2O) emissions applied to a case study of slurry management of swine production in southern Brazil. The manure management system (MMS) defined was the slurry tank without a natural crust cover. The calculation procedures used were: (i) GES'TIM; (ii) Static mass flow model from Hutchings et al. (2013); (iii) Calculation procedure described by Hamelin et al. (2011); and three variations of the IPCC guideline (IPCC 2006): (iv) IPCC with European default parameters, IPCC (DF-EU); (v) IPCC with Latin American default parameters, IPCC (DF-LA); (vi) IPCC with parameters adjustment to represent Brazilian reality, IPCC (BR). Finally, it was proposed a model guide for future estimations of MMS emissions in Brazil for slurry tanks (Baseline). Results showed significant differences between the upper and lower CH_4 and N_2O estimative.

Keywords: Emissions, calculation procedure, emission factor, CH_4, N_2O.

1. Introduction

Life cycle assessment (LCA) is a methodology widely used to predict the environmental profile of livestock and other agricultural products. Studies from literature (Basset-Mens; van der Werf 2005; Flysjö et al. 2011; Reckmann et al. 2013; Prudêncio da Silva et al. 2010; Prudêncio da Silva et al. 2014; Williams et al. 2006) have shown that the environmental impacts of these activities are mainly due to ammonia (NH_3), nitrous oxide (N_2O), carbon dioxide (CO_2), nitrate (NO_3) and methane (CH_4) emissions from crops cultivation by fertilizer usage, enteric fermentation from animal rearing, and manure management systems (MMS).

Regarding to swine production, manure handling is an important stage to mitigate the environmental impacts of this activity. According to Chadwick et al. (2011), the MMS selected by the farmers has a direct influence on the magnitude of gaseous losses and consequently the potential to reduce those emissions. For climate change, MMS represents 18-26% of total CO_2 eq. emissions (Kool et al 2009; Nguyen et al. 2011) which highlights the importance of having reliable data for CH_4 and N_2O emissions.

However, these emissions are difficult to be measured on field because of economic costs and the long period to measure it (Javon 2012). Therefore, many LCA studies uses mathematical models and emission factors (EF) to estimate CH_4, N_2O, NO_3 and NH_3 (Basset-Mens et al. 2007; Dalgaard et al. 2008; Flysjö et al. 2011; Nguyen et al. 2010; Prudêncio da Silva et al. 2014; Ruviaro et al. 2014; ten Hoeve et al. 2014; Wesnæs et al. 2009). There are a few calculations procedures and EFs in literature to estimate these emissions (Deltour et al. 2009; Gac et al. 2007; Hamelin et al. 2011; Hutchings et al. 2013; IPCC 2006; Rigolot et al. 2010), which may generate different results. Thus, it is not easy to choose the procedure and EF that better applies in the product system under analysis since these emissions have a high variability due to differences in system production, temperature, management, soil type, manure composition, windspeed and rainfall (Mkhabela et al. 2009, Sommer et al. 2009).

The aim of this paper was to perform a sensitivity analyzes of different calculation procedures and EFs to estimate CH_4 and N_2O emissions applied to a case study of slurry management of swine production in southern Brazil. In addition, we also proposed a calculation procedure for the Brazilian swine production based on the mathematical models and EFs reviewed.

2. Methods

The manure management system (MMS) defined was the slurry tank without a natural crust cover with a minimum storage period of 120 days with the land application of the stabilized manure (Fatma 2009). This system represents the most common MMS used in Brazil (Brazilian agroindustry; Higarashi et al. 2013; Kunz et al. 2005). The functional unit (FU) was the manure management generated to produce 1 ton of swine live weight, with the system boundaries set from the manure ex-house until its application on field. Only the emissions of CH_4 and N_2O produced from manure were considered. The characterization factors were according to IPCC (2007).

The calculation procedures used were: (i) GES'TIM (Deltour et al. 2009); (ii) Static mass flow model from Hutchings et al. (2013); (iii) Calculation procedure described by Hamelin et al. (2011); and three variations of the IPCC guideline (IPCC 2006): (iv) IPCC with European default parameters, IPCC (DF-EU); (v) IPCC with Latin American default parameters, IPCC (DF-LA); (vi) IPCC with parameters adjustment to represent Brazilian reality, IPCC (BR). Finally, it was proposed a model guide for future estimations of MMS emissions in Brazil for slurry tanks (Baseline). The input parameters for the mathematical models are shown in Table 1. We assumed a slurry tank without a natural crust cover, therefore we did not consider the direct N_2O emissions in storage (IPCC 2006).

Table 1. Input parameters for the mathematical models.

Parameters/Models	GES'TIM	Hutchings et al. (2013)	Hamelin et al. (2011)	IPCC (DF-EU)	IPCC (DF-LA)	IPCC (BR)	Baseline
Sows ($no \cdot FU^{-1}$)	0.34	0.34	0.34	0.34	0.34	0.34	0.34
Piglets ($no \cdot FU^{-1}$)	8.40	8.40	8.40	8.40	8.40	8.40	8.40
Swine ($no \cdot FU^{-1}$)	8.00	8.00	8.00	8.00	8.00	8.00	8.00
Sows ($days \cdot FU^{-1}$)	142	142	142	142	142	142	142
Piglets ($days \cdot FU^{-1}$)	38	38	38	38	38	38	38
Swine ($days \cdot FU^{-1}$)	112	112	112	112	112	112	112
$Manure_{sow}$ ($m^3 \cdot FU^{-1}$)	0.37	0.37	0.37	0.37	0.37	0.37	0.37
$Manure_{piglet}$ ($m^3 \cdot FU^{-1}$)	0.30	0.30	0.30	0.30	0.30	0.30	0.30
$Manure_{swine}$ ($m^3 \cdot FU^{-1}$)	4.00	4.00	4.00	4.00	4.00	4.00	4.00
Nex_{sow}^{b} ($kg\ N \cdot FU^{-1}$)	n.a	1.61	1.64	2.47	3.23	1.59	1.61
Nex_{piglet}^{b} ($kg\ N \cdot FU^{-1}$)	n.a	1.33	1.36	3.86	11.88	1.46	1.33
Nex_{swine}^{b} ($kg\ N \cdot FU^{-1}$)	n.a	17.49	17.89	57.12	175.84	17.96	17.49
VS_{sow}^{b} ($kg\ VS \cdot FU^{-1}$)	n.a	26.52	26.52	21.95	14.31	26.52	26.52
VS_{piglet}^{b} ($kg\ VS \cdot FU^{-1}$)	n.a	13.22	13.22	95.76	95.76	13.22	13.22
VS_{swine}^{b} ($kg\ VS \cdot FU^{-1}$)	n.a	174.23	174.23	268.80	268.80	174.23	174.23

[a] Not applied.
[b] VS = volatile solids; Nex = nitrogen excreted.

The methodological guide GES'TIM was elaborated to be reference for the quantification of gas emissions of livestock, soil, energy consumptions, inputs and compensation by carbon storage. The guide provides a homogenized methodological frame, with EFs that are representative of the French agricultural production sectors (Deltour et al. 2009). This method estimate CH_4 and N_2O emissions from manure storage and application in field. Detailed information on GES'TIM model can be found in Deltour et al. (2009), while input parameters considered in this paper are summarized in Table 2. Constant parameters and those that we assume as the same as it was described in the original calculation procedures were not added in this paper. For more details, see Deltour et al. (2009); Hutchings et al. (2013); Hamelin et al. (2011) and IPCC (2006).

In respect to GES'TIM model, we considered the same EFs for CH_4 and N_2O of Deltour et al. (2009) which implicitly assumes that the emissions from swine production in Brazil have the same EFs from swine production in France. These assumptions were also made by Prudêncio da Silva et al. (2014) for broiler chicken production. Therefore, to interpret our results, it should be in mind that there are some differences in the swine production in Brazil and France, such as: in France the manure is stored for some period in housing while in Brazil the manure goes directly to the slurry tanks which decreases the CH_4 emissions in housing (not considered in this paper) but could result in more emissions in storage since more volatile solids (VS) are available for decomposition. Another difference is the period of manure storage, i.e., in Brazil this period is 120 days while in France this period can be somewhat higher (Deltour et al. 2009).

The calculation procedure developed by Hutchings et al. (2013) describes the static mass flow of the manure nutrients, i.e. nitrogen and phosphorus (not considered in this paper) and the emissions of NH_3, N_2O, N_2, and NO_3. Therefore, to estimate CH_4 emissions we used the same model and parameters used in the IPCC (BR). In Hutchings model, the indirect N_2O emissions from NH_3-N and NO_x-N volatilization are not considered.

The calculation procedure from Hamelin et al. (2011) is based on IPCC (2006), however the authors included the estimative of N_2-N and NO_x-N and consider a different equation to estimate the NH_3-N loss in storage and manure application in field.

The scenarios following the IPCC guide differs from each other by the input parameters considered in the equation used to estimate the excreted N (Nex), and the VS content in manure and methane producing capacity (B_0) used to estimate CH_4 emissions. For IPCC (BR) different EFs for the N_2O emission in field and the N loss due to NH_3-N volatilization were also applied (Table 2).

Table 2. Input parameters to estimate CH_4 and N_2O.

Parameters/Models	GES'TIM	Hutchings et al. (2013)	Hamelin et al. (2011)	IPCC (DF-EU)	IPCC (DF-LA)	IPCC (BR)	Baseline
			Input parameters for CH_4 emissions in storage/field				
B_0 (m^3 CH_4 (kg VS excreted)$^{-1}$)	n.a[a]	0.29	0.29	0.45	0.29	0.29	0.29
MCF (kg·kg^{-1})	n.a	0.42	0.42	0.42	0.42	0.42	0.42
Conversion factor of m^3 CH_4 to kg CH_4 (kg·m^{-3})	n.a	0.67	0.67	0.67	0.67	0.67	0.67
EF_{CH4_St} (g CH_4·m^{-3}·d^{-1})	61.81	n.a	n.a	n.a	n.a	n.a	n.a
Storage period (days)	120	n.a	n.a	n.a	n.a	n.a	n.a
			Input parameters for N_2O emissions in storage/field				
E_1[b]	n.a	0	0	0	0	0	0
E_2[c]	n.a	0.085[d]	0.05[d]	0.48	0.48	0.05[d]	0.085[d]
E_3[e]	n.a	n.a	0.01	0.01	0.01	0.01	0.01
E_4[f]	n.a	0.256[g]	0.149[g]	0.48	0.48	0.05[d]	0.256[g]
E_5[h]	n.a	0.013[i]	0.013[i]	0.01	0.01	0.013[i]	0.013[i]
E_6[j]	n.a	n.a	0.0075	0.0075	0.0075	0.0075	0.0075
EF_{CH4_Field} (g CH_4·ha^{-1}·m^{-3}·d^{-1})	0.078	n.a	n.a	n.a	n.a	n.a	n.a
EF_{N2O_Field} (g N_2O·ha^{-1}·m^{-3}·d^{-1})	1.632	n.a	n.a	n.a	n.a	n.a	n.a
Application rate (m^3·ha^{-1})	38.8	n.a	n.a	n.a	n.a	n.a	n.a
$Area_{sow/piglet}$ (ha)	0.01	n.a	n.a	n.a	n.a	n.a	n.a
$Area_{swine}$ (ha)	0.14	n.a	n.a	n.a	n.a	n.a	n.a

[a] Not applied.
[b] Emission factor for direct N_2O emissions in storage without a natural crust cover (EF_3 in IPCC 2006, Table 10.21, chapter 10).
[c] Emission factor for NH_3 emissions in storage ($Frac_{GasMS}$ in IPCC 2006, Table 10.22, chapter 10 / NH3StoreRate in Hutchings et al. 2013 / NH_3-N in Hamelin et al. 2011).
[d] Values from Basset-Mens; van der Werf (2005), 0.05 kg NH_3-N (kg N)$^{-1}$ or 0.085 kg NH_3-N (kg TAN)$^{-1}$.
[e] Emission factor for indirect (NH_3-N+NO_x-N) N_2O emissions in storage/field (EF_4 in IPCC 2006, Table 11.3, chapter 11).
[f] Emission factor for NH_3 emissions in field ($Frac_{lossMS}$ in IPCC 2006, Table 10.23, chapter 10 / NH3FieldRate in Hutchings et al. 2013 / NH_3-N in Hamelin et al. 2011).
[g] Values from Basso (2003), Basso et al. (2004), 0.149 kg NH_3-N (kg N)$^{-1}$ or 0.256 kg NH_3-N (kg TAN)$^{-1}$.
[h] Emission factor for direct N_2O emissions in field (EF_1 in IPCC 2006, Table 11.1, chapter 11).
[i] Values from Gonzatto (2012), 0.013 kg N_2O-N (kg N)$^{-1}$.
[j] Emission factor for indirect (NO_3 leaching) N_2O emissions in field (EF_5 in IPCC 2006, Table 11.3, chapter 11).

3. Results

The results showed that IPCC (DF-LA) has the highest CO_2 eq. emissions while lower emissions were estimated through the calculation procedure described by Hutchings et al. (2013), see Table 3 and Figure 1.

Table 3. Results per FU (percentage of the contribution per column are given in parentheses).

Emissions per source/Models	GES'TIM	Hutchings et al. (2013)	Hamelin et al. (2011)	IPCC (DF-EU)	IPCC (DF-LA)	IPCC (BR)	Baseline
Emissions in storage (in kg CO_2 eq.)							
CH_4	865.3 (82.7)	436.5 (81.7)	436.5 (69.3)	1223.6 (77.1)	773.0 (41.5)	436.5(73.0)	436.5 (75.6)
direct N_2O	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
indirect (NH_3-N+NO_x-N) N_2O	n.c[a]	n.c	4.9 (0.8)	142.6 (9.0)	429.2 (23.0)	4.9 (0.8)	5.5 (0.9)
Emissions in field (in kg CO_2 eq.)							
CH_4	0.1 (0.0)	n.c	n.c	n.c	n.c	n.c	n.c
direct N_2O	181.6 (17.3)	97.7 (18.3)	120.8 (19.2)	154.5 (9.7)	465.0 (24.9)	121.5 (20.3)	97.7 (16.9)
indirect (NH_3+Nox) N_2O	n.c	n.c	51.5 (8.2)	30.9 (1.9)	93.0 (5.0)	13.9 (2.3)	15.2 (2.6)
indirect (NO_3 leaching) N_2O	n.c	n.c	16.0 (2.5)	34.8 (2.2)	104.6 (5.6)	21.0 (3.5)	22.3 (3.9)
Total	**1046.9 (100.0)**	**534.2 (100.0)**	**629.7 (100.0)**	**1586.4 (100.0)**	**1864.8 (100.0)**	**597.9 (100.0)**	**577.1 (100.0)**

[a] Not considered.

CH_4 emissions in storage contributed with 41.5-82.7% of total CO_2 eq. Indirect N_2O emissions in storage had a contribution of 0.8-23.0% of total CO_2 eq. Only in GES'TIM model the CH_4 emissions due to manure application in field were estimated, however this emission had minor contributions (0.005%). Direct N_2O emissions in manure application represented 9.7-24.9% while the indirect N_2O emissions (i.e. from volatilization of NH_3-N+NO_X-N and NO_3 leaching) participated with 1.9-8.2% of total emissions.

4. Discussion

Higher climate change potential was observed in IPCC (DF-LA), as displayed in Figure 1, greater CO_2 eq. emissions for this calculation procedure was mainly due to direct and indirect N_2O emissions. This occurred because in this model the default values for N excretion rate used in Equation 10.30 from IPCC (2006) are very high, which results in higher values for the excreted N (Nex in Table 1) when compared to the other models. Comparing to the default values for Western European the N rate for Latin American countries are 3 and 1.3 times (values not showed) higher for the market and breeding swine, respectively. The greater amount of Nex in IPCC (DF-LA) were also the main responsible for higher indirect N_2O emissions due to NO_3 leaching in this model. Another issue that resulted in greater indirect N_2O emissions in IPCC (DF-LA) and IPCC (DF-EU) was the default values for N loss due to volatilization of NH_3-N and NO_x-N considered in IPCC (2006). Ammonia and the nitrogen oxides emissions have greater influence in the results due to the nutrient balance and the indirect N_2O emissions

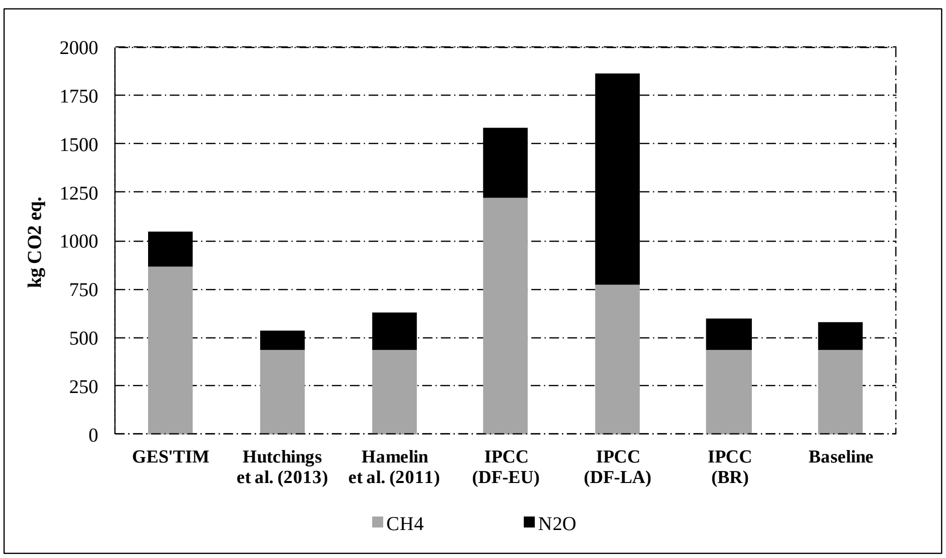

Figure 1. Comparison of mathematical models for CH_4 and N_2O emissions in manure management.

The higher CH_4 emissions in storage stage for IPCC (DF-EU) (Figure 1), compared to the other models, were mainly due to the methane producing capacity (B_0 in Table 1). Most probably, this is also the reason for greater amounts of CH_4 emitted in GES'TIM model. The B_0 for Latin American countries is lower than for Western European (IPCC 2006). Although we assume the same B_0 (i.e. 0.29 m^3 CH_4 (kg VS excreted)$^{-1}$) in IPCC (DF-LA), Hutchings et al. (2013), Hamelin et al. (2011), IPCC (BR) and Baseline models, IPCC (DF-LA) had higher CH_4 emissions due to the VS content used as input parameter.

Regarding the proposed model to estimate CH_4 and N_2O emissions from the MMS in Brazil, it can be noticed a need for a more site-specific data for the EF used for direct N_2O emissions in storage stage and for the constant input parameters used to estimate the proportion of organic N mineralized in slurry tanks and the coefficient factor used for NO_3 leaching emissions.

5. Conclusion

The results showed great differences for both emissions per FU according to the calculation procedure used. The model with lower CO_2 eq. emissions was from Hutchings et al. (2013), it showed a reduction of 71.4% when compared to the IPCC (DF-LA). CH_4 emissions in Baseline, IPCC (BR), Hutchings et al. (2013), and Hamelin et al. (2011) were 64.3% lower than in IPCC (DF-EU). While for N_2O emissions, the calculation procedure described by Hutchings et al. (2013) represented a 91.1% of reduction when compared to IPCC (DF-LA). The range in terms of CO_2 eq. emissions for CH_4 was 436.5-1223.6 kg while for N_2O was 97.7-1091.8 kg.

For CH_4 emissions, the differences were due to the methane producing capacity (B_0) and VS content. For N_2O the amount of excreted N considered and the N loss due to volatilization of NH_3-N and NO_x-N were the main reasons that significantly increased these emissions in IPCC (DF-EU) and IPCC (DF-LA).

Analyzing the models and based on our judgment, the calculation procedure used in Hutchings et al. (2013) seemed to be the preferable because it considers the effect of the mineralized organic N in the emissions, allowing future comparisons with other MMS, such as composting and biodigester. We suggest that future LCA studies also consider different options to estimate the CH_4 and N_2O emissions. Regarding to the Brazilian model proposed in this study, there is a need to develop site-specific data.

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The potential for reducing greenhouse gas emissions from health care via diet change in the U.S.

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ABSTRACT

We created three model healthy alternative diets (HADs) for the US based on USDA recommendations and compared them to the standard American diet (SAD). We estimated the relative risk (RR) for changes in consumption of the foods for three non-communicable diseases based on published meta-analyses. We then calculated the changes in health care costs resulting from reduced RR with the change from SAD to HADs, and the changes in downstream greenhouse gas emissions (GHGE) attributable to these costs. We found significant reductions in disease, health care costs and GHGE. Because we were conservative in the degree of diet change, and in selecting only the highest quality data on disease risk for foods, these results likely underestimate the total potential of diet change to mitigate GHGE. Significantly greater GHGE mitigation is anticipated from larger changes in diet and inclusion of more food-disease risk reductions. In addition, upstream mitigation of GHGE from HADs from changes in the agrifood system will be larger than those presented here for downstream effects—these estimates are included in the larger project on the potential contribution of diet change to mitigating climate change, for which this paper develops a key methodological component.

Keywords: Climate, diet, health care costs, non-communicable diseases, nutrition

1. Introduction

The composition of the diet has a substantial impact on our health and the climate. During recent decades the composition of the standard American diet (SAD) in the US has become markedly less healthy, and these changes, in combination with an increasingly sedentary lifestyle, have resulted in an epidemic of non-communicable diseases (NCDs)(Grotto and Zied 2010). In the US, 35% of the adult population suffers from cardiovascular disease (Go et al. 2014), 9.3% has diabetes(CDC 2014), and 40% is estimated to be diagnosed with cancer during their lifetime (SEER NCI 2014). The epidemic of non-communicable diseases is an important contributor to increasing U.S. health care costs to almost $3 trillion per year, representing 18% of the total US GDP in 2014, and 20% by 2022 (CMS 2013:Table 1). The toll of these diseases can be greatly reduced by adopting a healthy lifestyle, including healthy diets (WCRF/AICR 2007, WHO/FAO 2003).

Our overall goal in this paper is to develop a methodology for assessing downstream mitigation potential, and to use it to estimate the effect on greenhouse gas emissions (GHGE) from the health benefits associated with the adoption of a moderately healthier counterfactual diets by the US population based on USDA recommendations. Our results significantly underestimate potential diet change emissions mitigation potential, because they are based on a limited number of diet change and food-disease links, and do not included upstream GHGE. The research reported here will be expanded in scope to include these aspects, as part of a larger project.

2. Methods

The boundaries of our study were the US, although data for diseased risk were from various countries. The reference year was 2013, and we adjusted data to 2013 based on trends or indices. The system boundaries were the stages of the health care sector associated with the studied diet related diseases.

We estimated the effect of dietary change on GHGE, NCDs, and health care costs in four steps. In step 1, we defined the reference diets, the SAD loss-adjusted food availability at the consumer level in the US (USDA ERS 2014), and healthy alternative diets (HADs). In step 2, we estimated the changes in disease prevalence from dietary change, based on RR (relative risk) estimates found in the literature. In step 3, we estimated changes in health care expenditures from dietary change, based on changes in disease risk from step 3, using the most recent reliable expenditure data. In step 4, we calculated changes in GHGE from changes in health care costs from step

3, using data in the Economic Input-Output Life Cycle Assessment (IO-LCA) based at Carnegie Mellon University (GDI 2014).

2.1. Step 1. Developing dietary scenarios

To analyze the effect of dietary change on health and GHGE in the US, we used as a reference the SAD and compared it with three counterfactual healthy alternative diets (HAD-1, -2, and -3) (Table 1). Dietary intake levels in SAD were based on per capita loss-adjusted food availability data by weight for 2012 (USDA ERS 2014). These data estimate the average actual food intake in the US in cooked weights, based on amounts available at farm gate, adjusted for losses from farm gate through post consumer stages (Muth et al. 2011). In order to distinguish between unprocessed and processed meat, and between whole grains and refined grains which are aggregated in the data provided by the USDA, we assumed that consumption of processed meat accounted for 22% of total meat intake (Daniel et al. 2011), and consisted of 90% red meat, and that consumption of whole grains and refined grains accounted for 90% and 10% of total grain consumption, respectively (Lin and Yen 2007, USDA 2010).

Table 1. Per capita intake[1] of reference (SAD) & model (HAD) diets (g day^{-1}).

Food	SAD	HAD-1	HAD-2	HAD-3
Red & processed meat	92	51	25	0
Fruits & vegetables	328	657	657	657
Beans and peas	7	15	50	84
Total grains	167	131	131	131
Whole grains	17	79	79	79
Refined grains	150	52	52	52

[1]Intake levels in cooked weights. Basis for RR calculations.

To create the HADs, we adjusted SAD only for foods for which (i) USDA dietary recommendations were consistent with international nutrition and health authorities (USDA 2010, WCRF/AICR 2007), (ii) there were documented GHGE synergies, and (iii) there were high quality data on contribution to disease (section 2.2). The dietary recommendations for which there are documented GHGE synergies were: (i) eat no more calories than needed to maintain a healthy body weight, (ii) increase the proportion of calories coming from plant-based food, iii) reduce the consumption of meat (especially coming from red and processed meat), and iv) reduce the consumption of foods with low nutritional value (Garnett 2011). Creation of the HAD diets thus involved only a portion of the total SAD diet; we did not change any other food groups (e.g. unprocessed white meat, fish, dairy, eggs).

In HAD-1 we increased the intake of fruits, vegetables and whole grains, and reduced the intake of red and processed meat and refined grains from the levels in the SAD to the USDA recommended levels (USDA 2010). Processed meat was limited to 10 g of cooked meat per day based on the recommendation by the WCRF that processed meat should be avoided or limited as much as possible (WCRF/AICR 2007). We assumed that whole grains and refined grains contributed to 60% and 40% of total grain intake, respectively, based on the USDA recommendation that at least half of the grain consumption should come from whole grains (USDA 2010). We limited fruit juice to 20% of total fruit consumption based on the USDA recommendation that the majority of fruit intake should come from whole fruits (USDA 2010). By using whole food-based recommendations (e.g. vegetables), as opposed to nutrient-based recommendations (e.g. fiber), we reduced the risk of double counting health effects from nutrients found in various food groups.

We converted recommended food consumption levels provided by the USDA (USDA 2010) into grams day^{-1} using serving size weights given in (USDA ERS 2014). According to these data, one ounce equivalent of meat and grains equals 28.3 g (cooked weight) and 22 g, respectively (USDA 2011); one cup of vegetables, beans and peas, and fruits including juices is equivalent to 123 g, 73 g (cooked weight) and 187 g, respectively).

Intake levels in HAD-2 and HAD-3 are the same as in HAD-1, except that consumption of red and processed meat was further reduced and replaced by an increased intake of beans and peas. For validation, the nutritional content of all dietary scenarios studied was estimated, and found to provide 3500-3900 kJ day^{-1}, i.e. about one third of the average recommended daily energy intake for adults (NCM 2004).

2.2. Step 2. Changes in disease prevalence with changes in diet

We based the selection of diseases to be included in this study on a literature review of epidemiological meta-analyses of research on the relationships between specific foods and diseases. The literature review was performed in the NCBI Pub Med database in March 2014 using as keywords the selected foods groups (e.g. "vegetables") and different NCDs (e.g. "coronary heart disease"). The diseases included in the review were coronary heat disease (CHD), hypertension, adult onset diabetes myelitis (AODM) and a range of cancers. We selected peer reviewed meta-analyses of prospective cohort and randomized controlled trial (RCT) studies, published between 2005 and 2014, that provided RR with 95% confidence intervals (CI). We judged the evidence for the diet-disease relationship as insufficient, probable, or convincing, depending on the RR estimates found and number of studies supporting the relationship: *convincing* if minimum two meta-analysis supported the relationship and if all meta-analysis located showed significant reductions in disease risk from the studied changes in diet; *probable* if a minimum of one meta-analysis located supported the relationship and if the most recently published meta-analyses with significant results showed a reduced risk from the studied changes in diet; and *insufficient* if the criteria for convincing and/or probable were not met. We chose estimates for reduced risk of disease conservatively by only including RR estimates where the evidence was convincing or probable, which limited the diseases studied to CHD, AODM and colorectal cancer (CRC).

The health effects of changing the diet from SAD to HAD were estimated by calculating a revised RR (RR_{re}) for each food-disease RR, assuming a log-linear dose response relationship between food intake and health outcome, as reported in the meta-analyses (Eq. 1):

$$RR_{re} = RR^{((x-y)/u)} \qquad \text{Eq. 1}$$

where RR is the original RR obtained from meta-analyses for diet food f (e.g., processed meat) and disease d (e.g., CHD), x is the level of f in the HAD, y is the level of f in the SAD, and u is the unit increase reported in the meta-analysis identified for disease d. The reductions in RR for a unit change in food consumption were assumed to follow a uniform dose-response relationship across the range of intake levels in the SAD and HAD. When there was more than one meta-analysis RR for a food-disease combination, we used an arithmetic average of the *RR*s. For the relationship between whole grains and CHD, no dose-response RR estimates were located; therefore, a RR value based on the comparison of a high vs. low consumption was used to estimate this health effect. This was considered valid due to the large difference in intake levels of whole grains between the SAD and HADs.

We then calculated the combined effect of the changes in all of the foods contributing to the RR for each disease (RR_{cd}) by multiplying them, based on the assumption that the effect of each food was independent (Eq. 2) (Ezzati et al. 2006):

$$RR_{cd} = RR_{re1} * RR_{re2} * RR_{re3} * \ldots RR_{ref} \qquad \text{Eq. 2}$$

where RR_{re1}, RR_{re2}, RR_{re3}, and RR_{ref} are the recalculated RR values for each of the individual food changes in the diet. Finally, to construct the 95% confidence intervals around the relative risk estimates for the HAD we conducted a Monte Carlo simulation (Rubinstein 2007) with 5000 iterations in which the individual RR estimates were allowed to vary randomly according to a lognormal distribution.

2.3. Step 3. Changes in health care costs from changes in disease prevalence

The Medical Expenditure Panel Survey (MEPS 2011) is a standard source for health care cost data, but has widely recognized methodological limitations, so we used the most recent data for expenditures for the three diseases from alternative sources. Expenditures for CHD and CRC were from (Heidenreich et al. 2011, Mariotto et al. 2011), with spending category percentages assigned by percentages for all heart conditions by MEPS (AHRQ 2014). For AODM, which accounts for 90% of all forms of diabetes mellitus, we used (ADA 2013) for costs and spending category assignments, assuming that category expenditure distribution would remain constant.

Expenditures for each disease were then adjusted for inflation to 2013$. Because health care spending in the United States increases at a rate different than the standard rate of inflation, we used the Bureau of Labor Statistics consumer price index for medical care (BLS 2014) to adjust for inflation of medical expenditures.

2.4. Step 4. Change in GHGE due to changes in health care costs (ΔGHGE-H)

In order to estimate GHGE for health care expenses for each disease, we identified subcategories of expenses, since the types of services vary substantially (e.g., diabetes requires more prescription medications than heart disease). Subcategories were assigned in alignment with the relevant Carnegie-Mellon IO-LCA (GDI 2014) categories of medical expenditures: hospitals, pharmaceutical manufacturing, physician's offices, and home health services. CRC was assumed to have the same proportion of economic activity in those categories as all forms of cancers, CHD was assumed to have the same proportion of economic activity in categories assigned to all heart conditions, both taken from MEPS category spending assignments. The same method was used for AODM in the broader category of diabetes mellitus.

We then used the IO-LCA to determine an initial GHGE for each disease. However, because the IO-LCA uses CO_2e, or global warming potential (GWP), values from older IPCC assessments of various GHGs, we adjusted the GWP for CH_4 from 21 to the most recent IPCC GWP of 34 for a 100-year time frame, and 86 for a 20-year time frame (IPCC 2013:714, Table 8.7). We did this only for CH_4 because the N_2O GWP changed only a few percent.

Since the Carnegie-Mellon assessments were based on 2002 emissions levels, we adjusted for measured decrease in carbon intensity in the US economy from 2002-2011 (EIA 2013), and projected this to 2013. We assumed that the decrease in carbon intensity experienced by the US economy was the same as that in the health care sector.

Finally, we assumed that the RRre associated with HADs would result in a proportional decrease in expenditure.

3. Results

Changing from the SAD to HAD-1 reduced the RR of CHD, AODM and CRC by 20-40%, and HAD-2 and HAD-3 further reduced RR by 5-9%, respectively (Table 2, Fig. 1). The reduced RR of disease also resulted in an a 25-33% reduction in total health care costs for these diseases, and a total reduction in the US of 20-33 million MT CO_2e yr^{-1}, and 65-106 kg CO_2e $person^{-1}$ yr^{-1}.

Table 2. Reduction with change to HADs in disease, health care costs, and GHGE.

Diet change from SAD to	Reduction in									
	Relative risk of disease						Health care costs ($B yr^{-1}) (of $219.5 total for diseases)	Downstream GHGE (kg CO_2e $person^{-1}$ yr^{-1})		
	CHD		AODM		CRC			CH_4 GWP =21	CH_4 GWP =34	CH_4 GWP =86
	Combined effect	95% CI	Combined effect	95% CI	Combined effect	95% CI				
HAD-1	40%	29-51	35%	28-44	20%	13-26	54	64.5	69.0	87.0
HAD-2	45%	31-67	41%	32-50	25%	17-32	65	74.5	79.7	100.5
HAD-3	45%	32-58	43%	34-53	29%	20-37	72	78.5	84.0	105.8

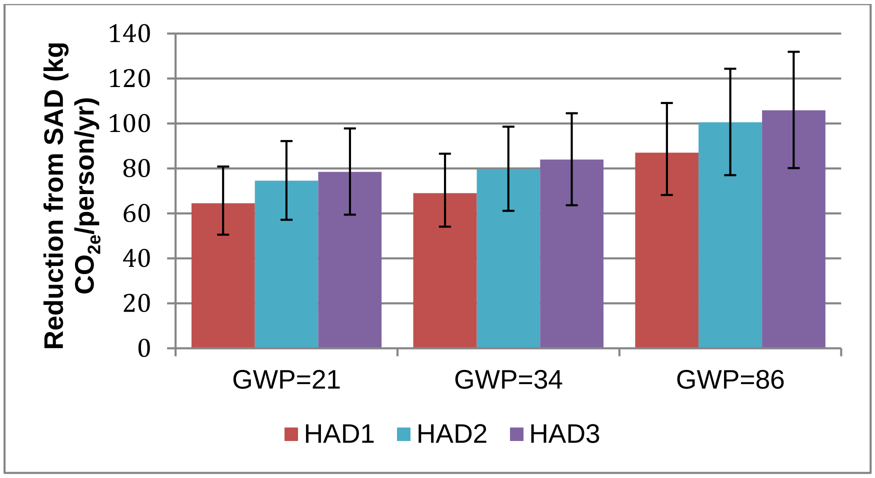

Fig. 1. Downstream reduction in CO_2e person^{-1} yr^{-1} for a CH_4 GWP of 21, 34, and 86.

4. Discussion

4.1. Results in relation to previous studies

Research on the combined effects of diet on the environment and nutrition is new but expanding. In recent years nutritional aspects have increasingly been incorporated into environmental and LCA assessments of food and diets, however, so far this has only included epidemiology to a limited extent. A small number of studies have examined the effect of healthier counterfactual diets on reduced morbidity and mortality, and on GHGE upstream (Aston et al. 2012, Scarborough et al. 2012), and on the effects of diet on GHGE via improved health, for example, by linking increased body weight to increased GHGE from higher fuels energy use (Edwards and Roberts 2009). However, the link between diet, health and GHGE from associated health care costs which we estimated in this paper has to our knowledge not previously been attempted. This study also fills a geographical gap, as most research on GHGE from dietary scenarios is limited to European settings (Hallström 2013)

4.2. Limitations of the research

The results of this study are largely dependent on the assumptions underlying the data we used for food consumption, RR, health care expenses, and GHGE. Of special concern when combining RR estimates as in this study, is the risk of double counting. We designed the study to minimize the risk of double counting by only using RR-estimates coming from meta-analyses that adjusted for influencing confounders, such as other types of food intake, physical activity level and history of disease. Despite these efforts, the risk of double counting remains, meaning that the health effects from the studied dietary change may be overestimated.. The overall uncertainty in results was also estimated with Monte Carlo Analysis. However, other assumptions we made would have resulted in under estimation of GHGE reduction from our counterfactual diets (see below).

4.3. Policy implications

The downstream GHGE reduction for HADs is equivalent to removing 4.3-7.0 million automobiles from the US roadways; it is also significant compared with other mitigation measures, for example, 1.9-2.4% of the Obama Administration's goal of a 17% reduction below 2005 US emissions levels by 2020. When upstream GHGE reductions from the HADs are added these numbers will increase significantly.

Although our results showed substantial potential benefits from dietary change, the total potential of diet change to simultaneously improve health and reduce GHGE is likely underestimated because it only included only a small subset of the possible foods and food-related NCDs. The total assigned expenditures for the diseases studied amounted to only \$220 billion, less than 8% of the projected total health care spending for the US in 2013 od \$2.9 trillion (CMS 2013:Table 1). Therefore, the savings we found from switching to HADS of \$B 76-94 yr^{-1} are likely a small portion of the potential savings, given that many more diseases and conditions are associated with diet. These savings (assuming consumer rebound is not significant), could be used for programs to support diet change and to retrain workers whose jobs would disappear as a result of the macroeconomic benefits of diet change.

But how realistic are the behavior changes required for the HADs? For HAD-1, the intake of unprocessed red meat and processed meat is reduced by 17 grams and 24 grams, respectively. The required change in meat consumption corresponds to eating one quarter pounder less per week and 1.5 sausages per week instead of 5.5. In order to reach the intake levels of fruits and vegetables and whole grains in HAD-1, the consumption in the SAD would have to increase by 2 and more than 4 times, respectively. Regardless of actual behavioral changes that could result over the coming decades, the attribution of GHGE to current behaviors remains a significant policy challenge.

In HAD-2 and -3 we decreased and eliminated red and processed meat. Lower intake levels of meat are also not a concern from a health perspective as long as reductions in meat consumption are compensated for by increased intake of other foods. Guidelines for healthy lacto-ovo-vegetarian and vegan diets are, for example, provided by the USDA (USDA 2010). In fact, there is increasing evidence that animal foods are a significant risk factor for many diseases (e.g. Farvid et al. 2014, Jakobsen et al. 2009, Kim et al. 2013), and also contribute disproportionately to the GHGE of the food system (Garnett 2009).

5. Conclusion

There is growing awareness of the importance of diet in determining GHGE and health care costs. We have reported here a methodology for quantifying this relationship. Our results show that it is possible to estimate the effects of changing to healthier diets with a high level of probability for the small proportion of foods and related diseases for which adequate data exist. The effects are significant in terms of improved health, reduced costs, and GHGE reductions. The methodology developed here is a key component of our larger project to estimate the net GHGE mitigation potential (from agrifood system change to health care) of diet change for a range of diets as alternatives to the SAD. Given the urgency of mitigating emissions over the short term in order to avoid catastrophic climate change, the mitigation potential of diet change should be investigated more thoroughly for incorporation into national, state and local climate policies.

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Challenges of Scale and Specificity in Greenhouse Gas Calculators

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ABSTRACT

LCA tools to certify products and guide farm management have proliferated but a detailed comparison is needed to show whether different tools are consistent. In this work, three widely used tools for cradle-to-gate emissions of greenhouse gases (GHGs) have been investigated - one general tool (Cool Farm Tool) and two crop-specific tools (Bonsucro for cane sugar and PalmGHG for palm oil). Large discrepancies are found even when the same input data are used; they result from detailed model differences so that no individual tool can be identified as "best" or "preferred". The results raise questions over the extent to which comparability can be expected between tools, particularly when they differ in their specificity to particular geographical areas and crops, and whether farm-level calculations can "add up" to provide estimates for GHG emissions at landscape or national level. Furthermore, generic tools may not be sufficiently representative to guide farm management for specific crops. Agreement is needed on how to reconcile results between different tools and levels.

Keywords: Greenhouse gas calculators, carbon in supply chains, farm management, standardization, comparability

1. Introduction

LCA is a scientifically-based analysis, but differs from conventional scientific models in not being amenable to empirical testing. In this respect, LCA has much in common with economic models. The inherent uncertainty in LCA models has led some to assert that even consequential LCA "cannot produce definitive quantitative estimates of actual environmental outcomes" (Plevin et al. 2014) and is therefore of limited value for policy support, even though similarly uncertain economic models are widely used for these purposes. Where LCA is used for certification or comparative claims, standardisation of approaches is therefore essential, even for the kind of attributional analysis which is used for product labelling (Clift et al. 2009).

Greenhouse gas (GHG) emissions from agriculture and forestry are responsible for nearly 30% of global anthropogenic emissions (IPCC 2007). Therefore emissions from agricultural production have attracted political, media, academic, and public attention at both 'macro' (i.e. global and national) level and 'micro' level (i.e. focusing on the individual farm), where the terms 'macro' and 'micro' are used here in their conventional economic senses. Variability and resultant uncertainty are problems for both LCA and economic models of agricultural systems, particularly in comparison to industrial processes whose performance is more uniform and predictable. In recent years there has been a proliferation of both consumer-facing agri-food certification schemes intended to encourage GHG management and measurement of crop production (Keller et al. 2013) and, at the 'micro' scale, of GHG calculators intended to report performance and aid management at the level of specific crops and individual farms (Colomb et al. 2013). However, there has been little attempt to compare and standardise calculators, which can yield different results even given the same input values for the same agricultural system (Colomb et al. 2013; Whittaker et al. 2013). This variability hinders creation of benchmark GHG figures or target ranges to guide producers and may limit their value in guiding on-farm management.

This paper compares three GHG calculators in detail: one general farm-focused calculator, the Cool Farm Tool (CFT) developed by the University of Aberdeen, Unilever and the Sustainable Food Lab (Hillier et al. 2011), and two crop-specific tools used by commodity roundtables, namely Bonsucro's GHG calculator for cane sugar developed at Louisiana State University (Macedo et al. 2008; Wang et al. 2008) and PalmGHG for palm oil developed by the Roundtable on Sustainable Palm Oil (RSPO) (Bessou et al. 2012 & 2014) (Chase et al. 2012). Both crop-specific tools have become central to the certification process (RSPO 2013; Bonsucro 2013). All three tools combine IPCC tier 1 national inventory data (IPCC 2006) with more regionalised tier 2 data, together with farm-specific activity data. Results are reported and compared here on a per land area basis (i.e. as kg CO2e per ha cultivated) to avoid including the additional variability due to the dependence of yield on very local conditions; for a specific farm, the impact per unit of crop is calculated using the actual site-specific yield. The results do not include the effects of specific management practices such as rotational cropping or tillage, nor the impacts of land use change. More detailed results and comparisons, including the effects of land use change, are given by Keller

(2014).

2. Methods

The three calculators have been applied to assess GHG emissions up to the farm gate, including agricultural production and associated inputs (e.g. fertilizer and pesticide production). Processing of the raw products is not included, nor are emissions embedded in infrastructure or equipment. The systems considered are sugar cane production in Brazil (Bonsucro and CFT) and palm oil cultivation in Indonesia (PalmGHG and CFT). Input data were provided by the relevant commodity round-tables (see Keller 2014) specifically for the purpose of comparing calculators; they are representative of common practice at the micro-level in the two cases but must not be interpreted as representing the sectors at the macro- or meso- levels (see Section 4). The tools differ in their data requirements and formats; e.g. only Cool Farm Tool requires information on soil properties. Minor adaptations to the calculators were made to enable them to use input data which were as near identical as possible but without modifying the underlying data or algorithms. Thus the results show the comparison between the tools in their generally available forms but with a few "bugs", identified in the course of this work, rectified in collaboration with the tool producers.

3. Results

The nature and extent of discrepancies between the general calculator and the relevant crop-specific tool are shown for sugar cane in Figure 1 and for palm oil in Figure 2. These Figures also reveal the GHG "hot spots" in the two systems: agrochemical inputs (including all soil or crop enhancers) and energy use (mainly diesel fuel for farm equipment and transport). Land use change is, of course, also highly significant but is not included in the comparisons presented here.

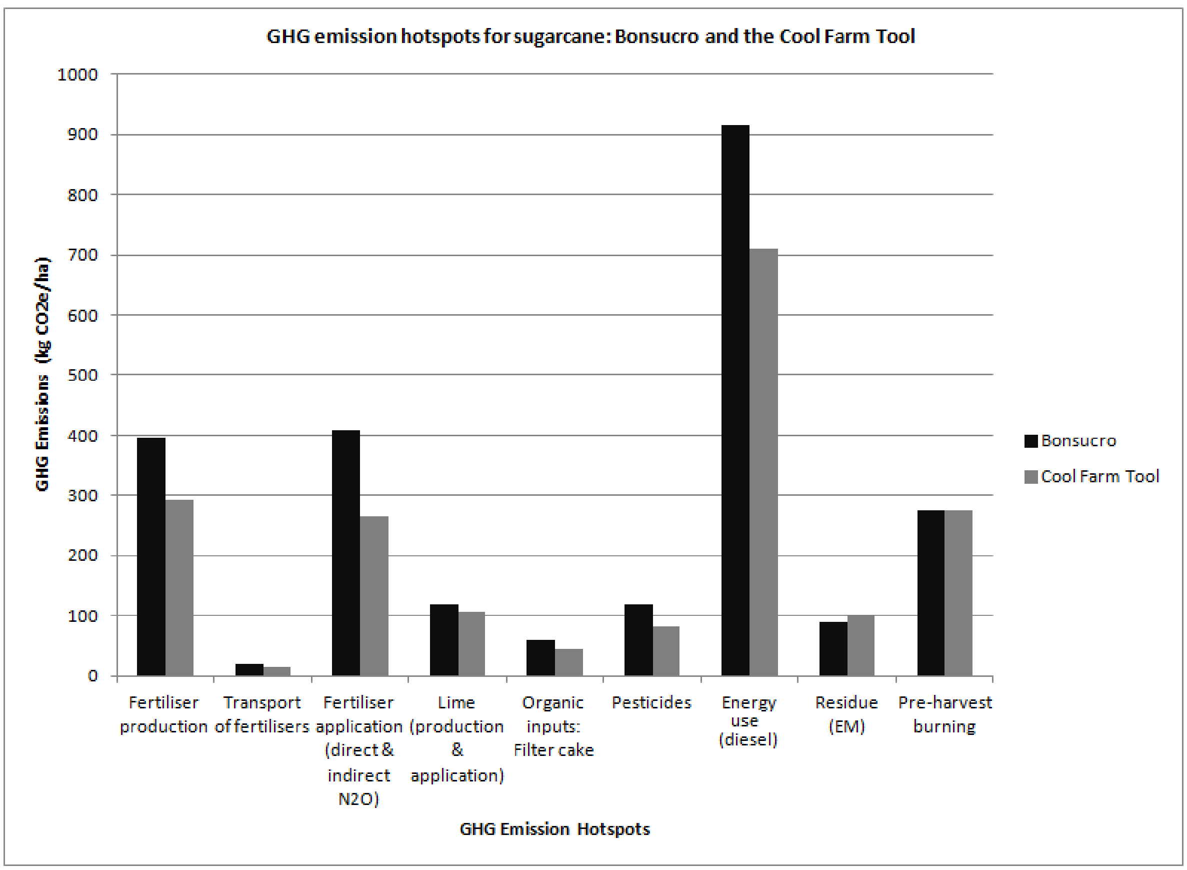

Figure 1. GHG emissions from sugarcane cultivation: Bonsucro and the Cool Farm Tool.

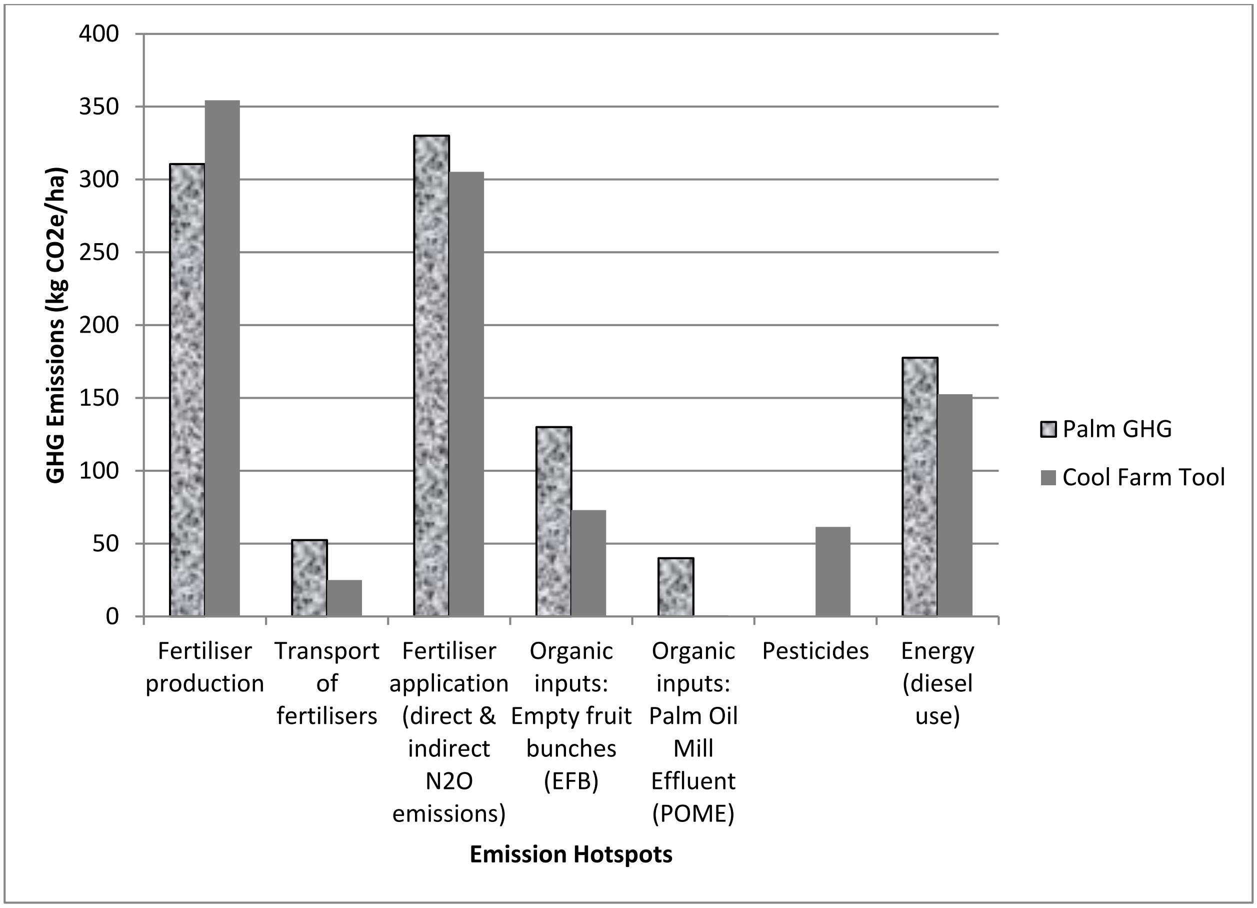

Figure 2. GHG emissions from palm oil production: Palm GHG and the Cool Farm Tool.

For almost all activities, CFT Gives lower estimates than the crop-specific tools. For energy use, the differences result from the emission factors assumed. The differences between the figures from fertilizer use are particularly marked for sugar - as high as 42% for emissions associated with mineral fertilizer. This also results from differences in the background estimates, in this case for emissions from production of nitrogen fertilizer: Bonsucro is based on data from the GREET model (Macedo et al. 2008; Wang et al. 2008) whereas CFT uses data for European fertilizer production that is generally more efficient. These differences illustrate a general problem in comparing calculators: even when they use the same input data and methodology, differences can arise from different background data and models; these aspects are explored by Keller (2014). Moreover, while Bonsucro and PalmGHG enable modeling of different inputs specified by the user, CFT as a multi-crop tool has more in-built flexibility and can model different practices such as reduced tilling and cover cropping. Thus CFT can be used to investigate improvements in management practices, a capability which is missing from Bonsucro and PalmGHG. Figure 3b compares results from Bonsucro with a range of different modes of fertilizer application calculated using CFT. Although the CFT results span a significant range, the range still lies consistently below the value calculated from Bonsucro. However, CFT estimates for different soil types (Figure 3a) span a wider range and do encompass the single estimate from the Bonsucro calculator. GHG emissions from fertilizer production for palm oil is the one component where CFT gives a higher estimate than the crop-specific tool but the range over different soil types is as wide as in Figure 3a so that the difference between PalmGHG and the baseline CFT estimates depends strongly on the soil type adopted as the default in CFT.

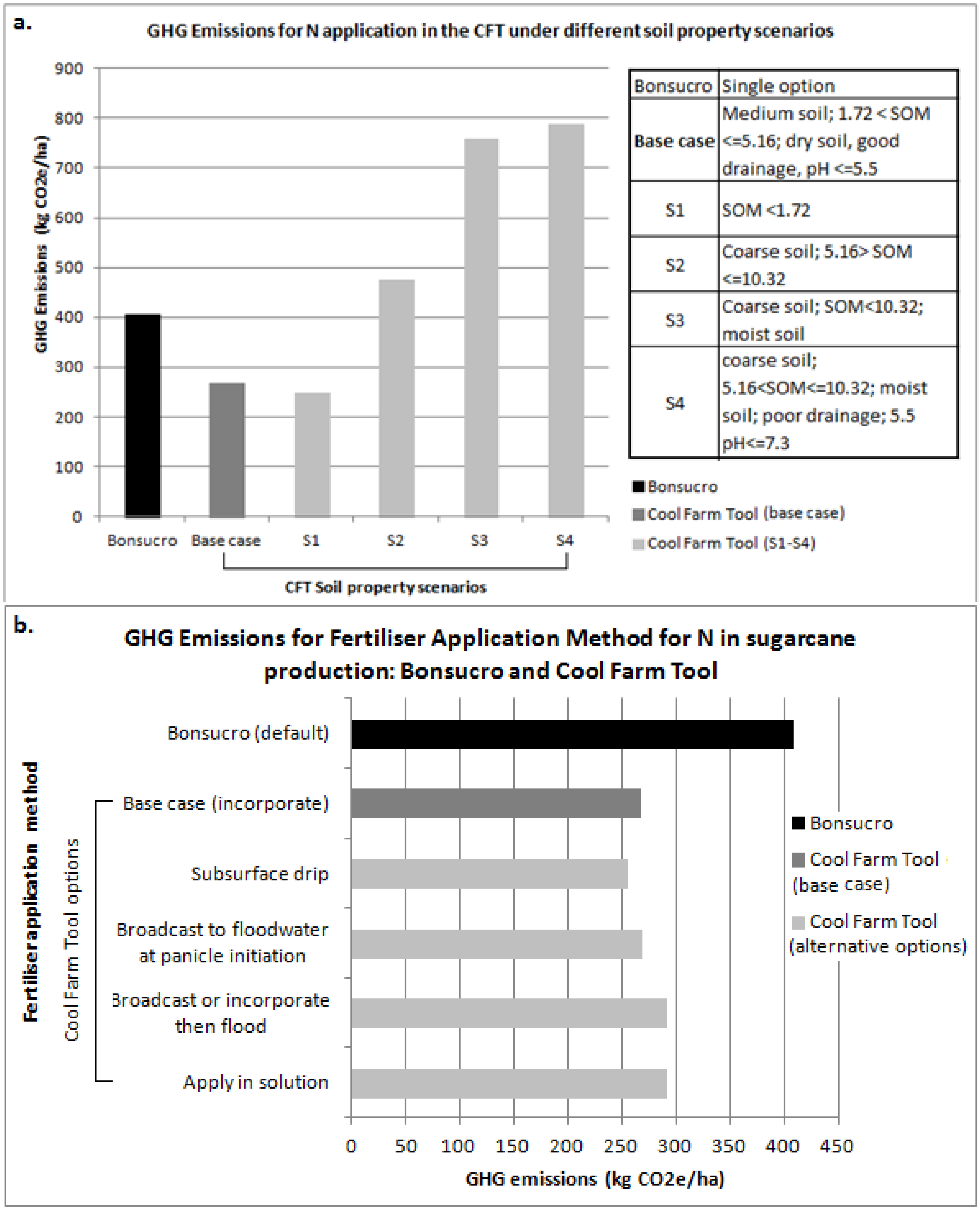

Figure 3. GHG emissions from fertilizer use in sugarcane cultivation: Bonsucro compared with a range of Cool Farm Tool scenarios for soil type and mode of application of fertilizer.
a. effect of soil properties
b. effect of fertilizer application method

4. Discussion

The results above show that, while these tools are in broad agreement, they give results which can differ by considerable margins. Short of detailed forensic examination of the underlying model data and algorithms, it is unrealistic to expect closer agreement. More critically, provided the tools are formulated correctly, there is no basis for prescribing any individual tool as the standard beyond ensuring that the most relevant background data are used. The differences between the results from different calculators are of comparable magnitude to the differences between individual producers; therefore comparative assessments must be based on results from a single calculator. This is particularly relevant to certification schemes (e.g. RSPO 2013) which require quantification of GHG emissions but permit the use of a specific tool (PalmGHG in this case) or an "equivalent" calculator but without guidance on what might be considered equivalent. However, we have not identified any respects in which the tools are contradictory in the sense of giving contradictory indications on ways to improve environmental performance at the farm level.

The discrepancies between these specific calculators raise the question of whether it will ever be possible to devise a single calculator for GHG emissions applicable to all crops and all purposes. In addition to the problems of describing different crops, production systems and farm or plantation management, there is an overarching question of the purpose for which the results are needed (Keller 2014). Going beyond the micro-/macro- distinction, it is useful to distinguish three different scales and applications:

Micro: products from a specific farm;

Meso: crops from similar agricultural systems;

Macro: global or national: relevant to a large area of land or landscape, with a number of different land uses and products.

Even within each level, there is a need for discussion aimed at agreement on appropriate methodology and data. GHG calculations at the micro-level are needed to guide farm management, and sometimes as the basis for certification or payments for responsible farm management; therefore they should be based as much as possible on primary farm-specific data with tier 2 or 3 data used for the background. Meso-level figures may be used, for example, to compare alternative food sources or the sustainability of different diets or as components of personal carbon calculators. Even within a single food supply or retail company, calculations at the two levels are used for different purposes: given the variability between different producers, micro-level calculations may be used to select individual suppliers and to help suppliers to improve their own performance, whereas meso-level calculations can guide what general products to promote and to select the regions from which they should be sourced. Macro-level figures are necessary for national GHG reporting, to support policy development and to support programs such as UN-REDD and REDD+. Tier 1 data are sufficient for this purpose and it could be appropriate to use environmentally-extended input/output calculations.

As usual with process-based LCA, calculations at the micro- and meso-levels do not necessarily add up to give the macro-level figures, for much the same reasons as LCA results for all a company's products may not add up to describe the whole of the company's impacts: the larger scale calculations include shared or background activities which need not be included if the purpose is to improve the environmental performance of a single farm, product or supply chain. Non-additivity should not be a concern provided that the three levels of calculator do not give perverse or contradictory results.

Some certification schemes are beginning to show attention to the landscape level, rather than focusing on individual crops. Rainforest Alliance, Fairtrade, UTZ along with other institutions and certification bodies have recently become members of the Committee On Sustainable Agriculture (COSA) seeking to analyze the impacts of agriculture at a larger scale, particularly associated with the implementation of sustainability initiatives (COSA, 2013). Bringing together and integrating performance information from certification schemes and other supply chain initiatives that may involve GHG calculators with domestic policies and larger programs including REDD could and should enable impacts to be assessed at the landscape scale (Nepstad et al. 2013) and ultimately scaled up to understand the global impact.

5. Conclusion

Even superficially similar "carbon calculator" tools can give considerably different results for identical cases and input data. At least for the tools compared here – Cool Farm Tool (CFT) vs. Bonsucro for sugar and vs. PalmGHG for palm oil – the differences are not attributable to the fact that CFT is a general tool whereas the others are crop-specific; rather, they derive from differences in the underlying models and, most importantly, in the background data which can only be revealed by very detailed forensic examination. It must therefore be anticipated that discrepancies between different calculators will persist. To achieve consistent and comparable results requires the application of consistent and comparable methods. Thus for comparative assessments for a single crop or level, it is essential to use one tool; comparative assessments or claims should be ignored unless they compare results obtained with the same tool.

Micro-, meso- and macro-scale assessments require different approaches. There is a need for discussion aimed at agreement on appropriate methodology and data for the different scales, although this will not remove the differences between different tools. Comparison across different sectors and scales is likely to remain problematic, dependent on understanding in depth of what lies beneath each calculator.

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AGRIBALYSE®, the French LCI Database for agricultural products: high quality data for producers and environmental labelling

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ABSTRACT

AGRIBALYSE is a French research program dedicated to producing Life Cycle Inventories (LCI) of agricultural products, based on a strong partnership of 14 research and technical institutes. It provides a homogenous and consensual LCI database to support environmental labelling policies and to help the agricultural sector to improve its practices. The public LCI database of more than 100 products and its detailed methodology report were published in 2014. The database mainly contains LCIs for average French products, the functional unit is the product mass (kg) and the perimeter cradle-to–farm-gate. The datasets go along with a detailed methodology report (Koch and Salou 2014) and a project report (Colomb et al. 2014). Several key objectives for a follow-up project have already been identified: better estimate of uncertainties, accounting for soil carbon dynamics, impacts on biodiversity and impacts of water consumption. Enlargement of the database with additional products and products from innovative systems is also demanded, as well as stronger connections with international programs. These issues should be addressed in a future AGRIBALYSE 2 program.

Keywords: LCI database, French agricultural products, ecolabelling, ecodesign

1. Introduction

AGRIBALYSE is a French research program (2010-2013) dedicated to producing public Life Cycle Inventories (LCI) of agricultural products. A database of more than 100 products and its detailed methodological report have been published in 2014 (Koch and Salou 2014).

AGRIBALYSE started in 2010, in response to the “Grenelle de l'Environnement” roundtables which decided on the use of Life Cycle Assessment (LCA) to support environmental labelling. The LCA approach was also seen as a tool to support farmers and their organisations to better identify the environmental impacts of their products and to support eco-design strategies to improve their production systems. At the same time, a lack of LCA results and especially homogenous ones was identified concerning French agricultural productions (Ecointesys 2008). Thus, AGRIBALYSE was launched by the French Environment and Energy Management Agency (ADEME). A strong partnership of 14 research and technical institutes (INRA, Agroscope, CIRAD, ACTA and 10 agricultural technical institutes) worked together to build a homogenous and consensual LCI database for French as well as a few imported agricultural products.

This article describes the main methodological choices and the general content of the database, whereas other contributions to the LCA Food 2014 conference describe more specific choices and results for each product sector (animal products/crops/fruits). The potential use and limits of the database will be discussed in the last paragraphs.

2. Methods: modelling framework and main methodological choices

For complete and detailed information on methodological choices, please refer to the AGRIBALYSE methodological report (Koch and Salou 2014).

2.1. General modelling framework and tools

In AGRIBALYSE, data describing agricultural practices have been provided by the technical institutes in an Excel file (called data collection tool), developed for the occasion. The data collection tool was then connected to models to calculate direct (on farm) emissions and resource use via another Excel file (called calculation chain). Indirect fluxes were then added in SimaPro, to obtain LCI and LCIA data. The characterization methods used were chosen based on the recommendation by the ILCD Handbook, the Envifood program and the ADEME-AFNOR platform (AFNOR 2011; JRC and IES 2011; ENVIFOOD 2012; GT1 2012).

2.2. System boundaries (space and time)

The perimeter considered in AGRIBALYSE LCIs is from cradle-to field-gate for crops and from cradle-to-farm-gate for animal productions. It implies that for crop production all the up-stream processes (production of inputs, field operations etc.) are included, but none of post-harvest operations, although they might happen on the farm (storage of potatoes, drying of cereals). All operations required for the animal production phase are included (feed production, transport and storage, fattening of animals, milking, construction and maintenance of buildings and machines).

For managing manure, the distinction between animal and plant production was defined according to Gac et al. (2010). Emissions from storage of manure and any forms of treatment (nitrogen reduction, composting or anaerobic digestion) were allocated to the livestock production system and the emissions associated with loading, transport and spreading were allocated to the plant production system which used the manure.

To be consistent, system boundaries have been described following 9 schemes (3 for plants, 6 for animals) (Figure 1), to cover all agricultural productions.

The reference period for LCIs was 2005 to 2009, except for perennial crops for which it was 2000-2010 to account for the production phases (seedlings, early production, full production etc.) and alternating yield phenomenon.

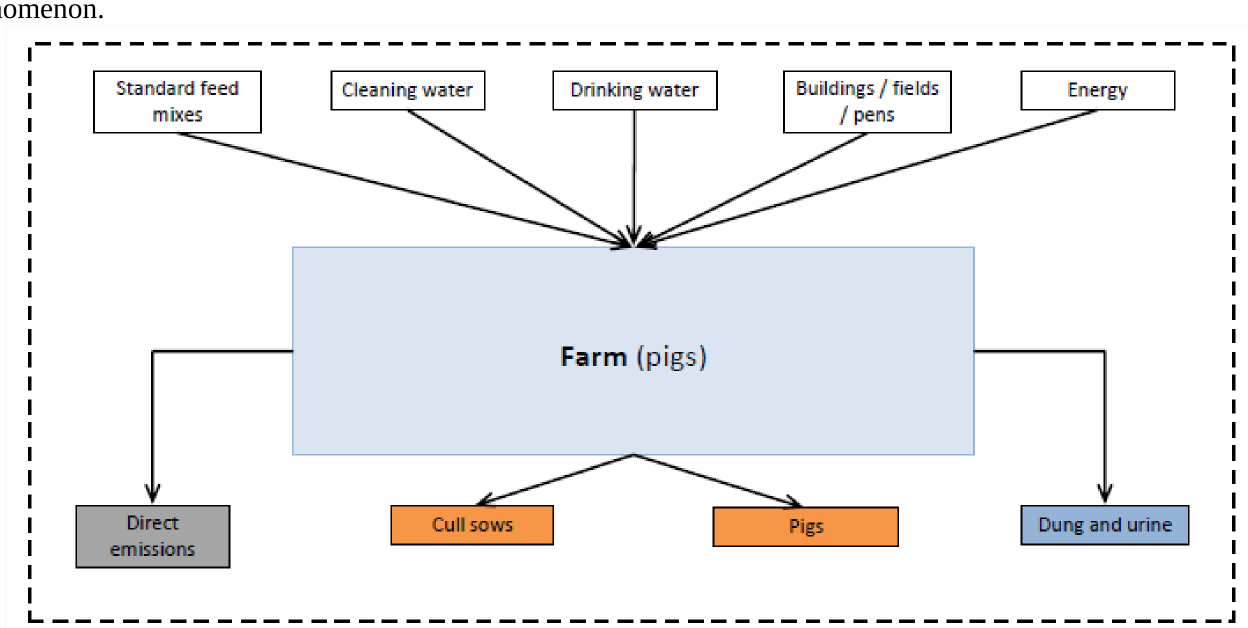

Figure 1. Example of system boundaries for pig production.

2.3. System description, data sources and representativeness

Initially, AGRIBALYSE aimed to provide a representative LCI for every major French agricultural product. The description of production systems by the technical institutes was based on (in order of preference): statistical sources, typical cases (documented in as much detail as possible), expert opinion or individual case / estimate as the last option. The quality of the description of the production systems was a key step for the quality of the database. For many products it was not realistic to identify a single production system to represent the French average. Therefore, several production systems for the same product (variations) were often described. E.g. for potatoes, the following LCIs were calculated: industrial use, fresh market firm flesh, fresh market other varieties, starch potato and a national average. The national average is a weighted average based on the share of each system in the national production. For production systems that count a large number of variations, the weighted average may not be representative of the national production (e.g the five dairy systems LCIs represent 21.1% of the national production of milk). For some products an average product was not meaningful (e.g. grapes) because each product is specific of a production system or an area of production (Beaujolais, Maconnais, etc.) and no national average could be provided. It is very important to be aware of the representativeness of each LCI when using the database.

2.4. Allocation

Allocation rules follow the international recommendations (ISO 2006a, 2006b). The allocation method must always be meaningful for all co-products implicated. For crop production, most of the co-products are generated in the processing step, which is not included in the AGRIBALYSE program. No allocation between different product qualities was applied at field-gate (e.g. export quality versus local market quality for imported fruits). For crops, an important work has been implemented to allocate organic N fertilizer and P and K mineral fertilizers between cropping sequences (Koch and Salou 2014, Annexe D). For animal production, a so-called bio-physical allocation method was implemented, based on Dollé and Gac (2012) for milk. At first, allocation was avoided by decomposing the system in animal classes based on their production phase. Then, for the animal classes where allocation was required (ex: dairy cow in production phase), allocation was based on the metabolic energy required to produce each co-product (calf, milk). The impacts of the animal classes producing a single product were fully allocated to this product. For data imported from pre-existing databases (eg. ecoinvent v2, INRA Rennes internal database), such as for animal feed components, the allocation choice made in the pre-existing database (mostly economic allocation) was not modified.

2.5. Models for the calculation of the direct emissions

AGRIBALYSE identified the most appropriate calculation methods for all direct emissions and resource consumptions, while pre-existing datasets (mainly ecoinvent v2) were used for calculating indirect emissions (up-stream ex: inputs production, material etc.).

Finding appropriate models to estimate direct emissions for each pollutant substance was one of the main challenges. According to the AGRIBALYSE set of rules, such models should not require input data that are too difficult to collect, should have been published, should have been validated for France (or tropical conditions for the tropical crops) and should be recognized internationally. The priority was given to inter-product consistency, with the idea of having specific models for each emission mechanism used across the entire database rather than having specific models for each product or type of product. The list of models used is available in the methodological report (Koch and Salou 2014).

2.6. Quality control

The quality control was performed in two steps: control of agricultural system data and control of LCI data (Figure 2). Quality control of agricultural system descriptions was performed by independent experts. LCI results were controlled by experts of the technical institutes involved in the program, both at the level of the calculation tools and regarding the plausibility of the results. More than 160 system description sets were checked with unqualified acceptance in 30% of cases, minor revisions in 50% of cases and major revisions in 20% of cases. More than 500 suggestions for improvement or comments on technical data and sources were taken into account to meet the recommendations of the experts. The quality control of the LCI results allowed the correction of calculation errors, the detection of anomalies, it ensured coherence and improved the credibility of the results. Performing a quality control on both the system description data and the LCI data sets significantly contributed to the quality of the AGRIBALYSE database.

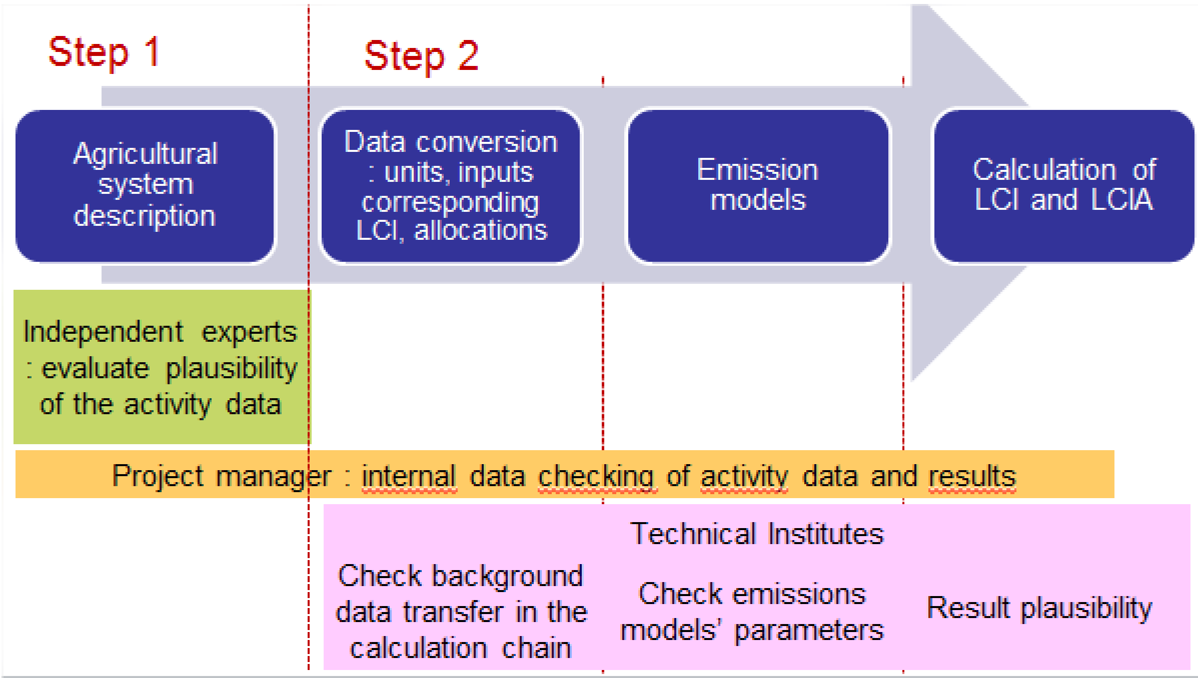

Figure 2: Quality control scheme

3. Results

The outcomes of AGRIBALYSE are a public LCI database, a methodological report (Koch and Salou 2014) and a final report describing the project organization and main lessons learnt (Colomb et al. 2014). The database is available in several formats with various levels of detail depending on the user's requirements:

o Factsheets: a simplified version in PDF format presenting most common and most robust impact indicators. The factsheets are freely available and do not require an LCA software.

o The AGRIBALYSE_vIMPACTS database (system processes, ILCD and Ecospold_V1 formats): containing aggregated LCI data sets considered by the AGRIBALYSE consortium to be sufficiently reliable to be used for a product environmental labelling approach (robustness, representativeness). These LCI data sets are available for incorporation into the ADEME IMPACTS® database, the official French environmental labelling database. An LCA software is required to analyse these data.

o The AGRIBALYSE_vcomplete database (unit processes, Ecospold_V1 format). These data are intended primarily for ecodesign projects and provides the results in a transparent format. An LCA software is required to analyse these data.

ILCD and ecospold formats are suitable for LCA specialists as the data can be used in current LCA software.

The reports and summary factsheets are available on line (www.ademe.fr/agribalyse-en), while the full database is available on demand to ADEME. Finally, a short document "Advices for proper use of AGRIBALYSE results" (AGRIBALYSE 2014) highlights the critical points when using and interpreting the data. The publication of data and the efforts made to ensure transparency, with several formats to respond to different needs has been an important time investment, but should contribute to a large and appropriate use of the datasets.

The products included in the database are 28 crop and 18 animal products (Table 1). For several products a number of variations exist.

Table 1 : list of AGRIBALYSE product groups and number of variants per product group.

Sector	**Type** (the product groups are given in brackets)	Number of product groups	Number of product variants
Arable / horticultural	**Annual crops** *(durum wheat, soft wheat, sugar beet, carrots, rapeseed, faba beans, grain maize, barley, peas, potatoes, sunflowers, triticale)*	12	28
	Grassland/forage *(grass, alfalfa, silage maize)*	3	16
	Fruit *(peaches/nectarines, apples, cider apples, wine grapes)*	4	13
	Special crops grown in France *(roses, tomatoes, ornamental shrubs[a])*	3	6
	Special tropical crops *(coffee, clementine, jasmine rice, cocoa, mango, oil palm fruit)*	6	6
Total	**Arable / horticultural**	**28**	**69**
Livestock	**Cattle** *(cow's milk, beef cattle, veal)*	3	14
	Sheep *(sheep's milk, lambs)*	2	2
	Goats *(goat's milk)*	1	1
	Poultry *(eggs, broilers, turkeys, ducks for roasting, ducks for foie gras)*	5	15
	Rabbits *(rabbits)*	1	1
	Aquaculture *(trout, sea bass / sea bream)*	3	3
	Pigs *(conventional, Label Rouge, organic)*	3	8
Total	**Livestock**	**18**	**44**

4. Discussion and perspectives

4.1. Improvement needs and perspectives

AGRIBALYSE proposes a comprehensive database and a detailed and coherent methodological framework. For further work several methodological improvements have been identified. The main issues to be resolved are the modelling of changes in soil and biomass carbon stocks (due to farmer practices and land use change), of impacts on biodiversity, and of impacts of water consumption.

The methods and models for calculating emissions (nitrate, phosphorus, trace metals, etc) may also be modified to take better account of the specific characteristics of crops, agricultural and animal production practices and soil-climate characteristics. For France, IPCC tiers 2 models for GHG emissions should be ready soon and could significantly modify the results. It will be therefore important to update the database and use the best models available.

The LCI database could also be enlarged, by including new products (ex: fishery products, other vegetables, more imported products, etc.) and new product variations (regional products, Protected Geographical Indication products, quality label products, etc.) could be included to strengthen the representativeness of the database. For a proper comparison of products, it would be necessary to better quantify the uncertainty of the LCI data. The quality of the data was assessed using the ecoinvent pedigree matrix and ILCD criteria. Data for tropical products and for less studied production systems (with less data available, like organic systems) hold the largest uncertainty.

The future AGRIBALYSE database will probably allow the inclusion of LCI data proposed by third parties. This would mean that a procedure should be established to examine whether the proposed LCI data have been produced in accordance with the AGRIBALYSE methodology and quality assurance standards. This will be a complex task that will have to be tackled. Finally, the database being based on ecoinvent v2 datasets, the relevance of updating to ecoinvent v3 should be assessed.

4.2. Contribution to Environmental labelling and Eco-conception

The LCI data needed for the implementation of environmental labelling will depend on the labelling scheme. The companies will need reliable background data (such as AGRIBALYSE) and tools to implement ecolabelling schemes.

By the end of 2013, a French parliamentary report on environmental footprinting highlighted the interest of consumers for environmental labelling of food products; it also identified several specific methodological issues for the food sector (Errante and Saddier, 2013). It advocates that, in France, environmental labelling should become a voluntary but regulated process in the future. The volunteer companies should apply a common methodology and communication framework, which would be defined by public institutes and would limit the risk of green-washing or incomplete analysis. No clear time frame for this development was given, but the will to implement global environmental labelling was reaffirmed. The European Union is also currently carrying out an experimentation on the Product Environmental Footprint (PEF) to determine the appropriateness and feasibility of implementing environmental labelling at the European scale. AGRIBALYSE data could be used for this experiment.

Thus the AGRIBALYSE LCI database is a key element for the environmental labelling of food products. It is however not sufficient as it does not include processing, marketing and end of life stages. The processing steps for food products are currently analysed in ACYVIA, another French LCI database program, that should provide results by the end of 2016. Both data from AGRIBALYSE and ACYVIA will feed ADEME "IMPACTS" LCI database, which should be the official multi-sectorial database supporting the French ecolabelling schemes.

AGRIBALYSE helps to guide the farming sector towards more ecoefficient agricultural systems. By providing a comprehensive method for future LCA studies of agricultural products, it will allow more comparable results and clearer conclusions. In April 2014, the database had been downloaded by more than 65 institutions (130 individual licences provided), research and technical institutes, agro-food companies, consultancies working on ecodesign. The data are also integrated in some LCA computer tools. The database is used for instance in French Ecoalim research project which tries to identify animal feeding strategies with lower environmental impact. Data have also been used for policy studies, for instance in a study for the Swiss government comparing the impacts of Swiss and imported products, in order to define an appropriate agricultural and environmental strategy. Last but not least, the LCI data can be used to explore the impacts of human diets.

4.3 AGRIBALYSE II

ADEME has decided to launch a follow up program for AGRIBALYSE. The general goals, providing data for environmental labelling and supporting ecodesign are still considered to be very relevant and will be conserved. The main goals for this program will be to (a) help users to produce LCI data following the AGRIBALYSE methodology, (b) update and maintain the data, (c) augment the database with LCIs of new key products, (d) promote and include methodological improvements from the LCA field.

Thus, AGRIBALYSE II should be a platform coordinating several thematic projects on dataset production and methodology improvements that would feed the database. The platform should be strongly connected to other European and international database projects.

5. Conclusion

The AGRIBALYSE partnership produced a consensual and homogenous LCI database for French farm products. By its ambition and innovativeness it also consisted in a major learning process for the partners involved. Nevertheless it must be stressed that the amount of work and skills required for such program has been considerable.

The database is now used by many French and international organisations: research and technical institutes, agro-food companies, consultancies etc.. These data are publicly available and can be used in many projects dealing with sustainable production system, eco-labelling, diet studies, agricultural policy strategies etc. For the future, several methodological issues remain to be implemented in the AGRIBALYSE methodology (dynamics of soil carbon stocks, water consumption, impacts on biodiversity) and the characterisation of uncertainty should be improved. Really understanding LCIA results requires understanding of the impact assessment methods including complex scientific models, thus efforts on communication means should be continued.

To conclude, we can say that AGRIBALYSE has been an excellent pilot program that now needs to be connected and enriched with other initiatives.

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Impact of transportation on the environmental performance of Brazilian banana production

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ABSTRACT

The increase of consumer's ecological awareness has forced decision-makers and producers to search for scientific information about environmental performance of agricultural production systems. LCA studies on tropical perennial agricultural products are still scarce; furthermore, the great diversity of these products contrasts with the reduced available environmental data in Brazil. As fifth largest banana producer in de world, Brazilian production is almost entirely directed towards the domestic market through small-scale banana producers widely distributed throughout the country. The objective of this study is to evaluate the environmental performance of banana transportation stage through environmental indicators, considering Brazilian largest production regions. In the domestic market most bananas are consumed near its production centers, except for Northern Minas Gerais and Bom Jesus da Lapa - BA regions which distances from the consumers are approx. 1,000 km. The banana consumed in São Paulo has the lowest environmental impact in relation to the transport stage: 23.20 kg CO_2 eq/t, 50.10 MJ/t and 8.41 kg of non-renewable resources/t in the empty return scenario. Exports impact was larger for MERCOSUR than Europe due to the use of different transportation modes.

Keywords: banana, Brazil, transport, environmental impact, GHG emissions

1. Introduction

The Brazilian banana crop in 2013 was 6,931,137 tons, occupied a harvested area of 523,315 hectares and had an average yield of 13.2 tons per hectare (IBGE, 2014a). In 2012, the value of the Brazilian banana production was estimated at 4,396 billion reais (IBGE, 2014b). Brazil features as the fifth largest producer of bananas in the world (FAO, 2012). However, Brazil is not even among the top ten exporters of the product, exporting 110,054 tons in 2011 (FAO, 2014), since its production is almost entirely directed to the domestic market. With the exception of the states of São Paulo, Paraná and Santa Catarina, where crops of Cavendish bananas prevail, most of the national production of bananas is from the subgroup *Prata*, preferred by most Brazilian consumers and more resistant to postharvest injuries and illness, but not well accepted in the export markets (Lichtemberg, Lichtemberg, 2011).

The ten municipalities that stand out in the national banana production account for less than 20% of the Brazilian production. This indicates the wide geographic distribution of small-scale banana producers throughout the country, with only a few poles that stand out as major producers, namely: North of Santa Catarina, the Vale do Ribeira in São Paulo´s Southern coast, North of Minas Gerais, Bom Jesus da Lapa in Bahia, Vale do Submédio do São Francisco, Vale do Açu in Rio Grande do Norte and Vale do Jaguaribe in Ceará. While states in the South and Southeast region export to MERCOSUR countries, especially for Argentina and Uruguay, the northeastern states, notably Rio Grande do Norte and Ceará, have had a growing market share in Europe (Germany, UK, Netherlands, Poland, Spain and Italy) (Vieira , 2009).

The increase of consumer's ecological awareness has forced decision-makers and producers to search for scientific information about environmental performance of agricultural production systems through Life Cycle Assessment (LCA) studies on food products. Nevertheless, additional work is required, as well as the development of a methodology to continue improving LCA studies on tropical perennial agricultural products, which great diversity contrasts with the reduced data, either because these data do not exist or because they have not been published.

A number of studies have reinforced the belief that the stage with the most significant impact for the banana industry is transportation, mainly because of the overseas transport. Craig et al. (2012) estimated the carbon footprint of bananas produced by Chiquita Brands International in Central and South America, and sold to the USA, at approximately 1 kg of CO_2eq/kg of bananas sold. The transportation stage represented the largest part (36%) of emissions by the supply chain, primarily due to overseas transport, followed by the stages of production (22%) and retail (22%).

Ecuador is the largest exporter of bananas in the world (FAO, 2014). Iriarte et al. (2014) analyzed the carbon footprint of Ecuador's Premium bananas for export (Musa AAA – *Cavendish*) using a considerable amount of field data (harvests of 2009, 2010 and 2011). The system boundaries considered from agricultural production to delivery in a European destination port, simulating two scenarios: the best-case scenario, where the refrigerated containers of the ships did not return empty from their trip to Europe and the worst-case scenario, where they returned empty. Accordingly, the carbon footprint of Ecuador's bananas for export ranged between 0.45 kg (best-case) and 1.04 kg CO_2-eq/kg of bananas (worst-case), as shown in Table 1. The study demonstrated the importance of using efficient transportation, i.e. the simple use of containers during the return stage makes possible to reduce the carbon footprint by 57%. The study concludes that the overseas transport stage has the highest contribution to the carbon footprint (27% to 67%) followed by agricultural production (23% to 53%).

Table 1. Contribution of life cycle stages to the carbon footprint of Ecuadorian export bananas (Iriarte et al., 2014).

Life Cycle Stage	**2009**	**2010**	**2011**
	(kg of CO_2eq/kg)	(kg of CO_2eq/kg)	(kg of CO_2eq/kg)
On farm	0.21	0.24	0.26
Post-harvest fruit handling	0.001	0.001	0.005
Packaging	0.08	0.09	0.09
National transport	0.008	0.008	0.008
Overseas transport[a]	0.12/0.71	0.12/0.70	0.12/0.70
Total carbon footprint[a]	0.42/1.01	0.46/1.04	0.48/1.06

[a] The first value shown represents the Best-case scenario for overseas transport while the second value shows the worst-case scenario.

Therefore, the aim of this study is to evaluate the environmental performance of banana transportation stage through environmental indicators, namely climate change, energy use and non-renewable resources, considering Brazilian largest production regions. Both domestic consumption and exports are taken into account in this study.

2. Methods

The LCA methodology employed was based on standards ISO 14040 and 14044 (2006). The modeling of transport data was carried out using Brazilian data on fuel consumption by the fleet and IPCC emission factors for GHG emissions (Ministério, 2011; IPCC, 2006). Partial life-cycle inventories related to fuel production and pre-combustion developed by CETEA - Packaging Technology Center (Coltro et al., 2003; Garcia et al., 1999) were incorporated into the transport inventory for bananas.

The goal and scope of this study was to evaluate the environmental impact of banana transportation inside and outside the country. The environmental indicators may be incorporated in LCA of bananas produced in Brazil, which could boost ecolabeling of this product and larger volumes of trade.

A broad survey on recent banana production data in Brazil characterizing it geographically, both domestic consumption and exports, was performed in trusted databases and literature. The information considered in this study (distances, fuel consumption, capacity of the transport modals, packaging etc.) were obtained via the collection of data from reliable sources. Extrapolations were justified based on data published in the literature.

The functional unity adopted was 1,000 kg of bananas delivered to retail in the domestic market or at port in case of export.

The system boundaries considered the fuel production and the transportation in the domestic market and exports to MERCOSUR and EUROPE, simulating three scenarios, as follow: 1) the ships and trucks returned empty from their trip, 2) they returned full, and 3) they returned full but with 33% loss due to injuries during the transportation.

Transport distances were estimated using Google Maps inserting the main producing cities in the region considered and the main consumption centers of its production. Exports to Europe considered Germany and the cities of Buenos Aires and Montevideo, when exported to Argentina and Uruguay respectively.

The considered impact categories are: climate change (CO_2 eq), energy use (MJ) and non-renewable resources (kg). GHG emission factors for transport and energy production were obtained from GHG Protocol Brasil (2012). The inventory quantities primary energy demand (PED) and non-renewable resources were analyzed. GWP (100 years) were estimated according to the CML method (GUINÉE, 2002).

3. Results

In Brazil, banana production is almost entirely directed towards the domestic market due to its large population and high per capita consumption, 29.1 kg of bananas per inhabitant per year (FAO, 2009). Accordingly, the country has not developed good postharvest handling and conservation practices for transportation to overseas markets, as have the more traditional banana exporting countries: Ecuador, Philippines, Colombia, Costa Rica, Guatemala etc. (Lichtemberg, Lichtemberg 2011). Small producers scattered over national territory are mostly responsible for Brazilian production.

In 2011, the domestic market consumed 98.5% of the banana production and a mere 1.5% was exported, primarily to Europe and MERCOSUR countries as seen in Figure 1 (IBGE, 2014b). In 2010, exports had been a little higher, reaching 2%. The main destinations for Brazil's exports in 2011 were Germany (31% of total value and 25% of the exported volume), Uruguay (25% of the total value exported), UK (16% of total value) and Argentina (14.5% of total value). In 2011, approx. 52% of the volume of banana exports was destined for MERCOSUR (SECEX, 2014).

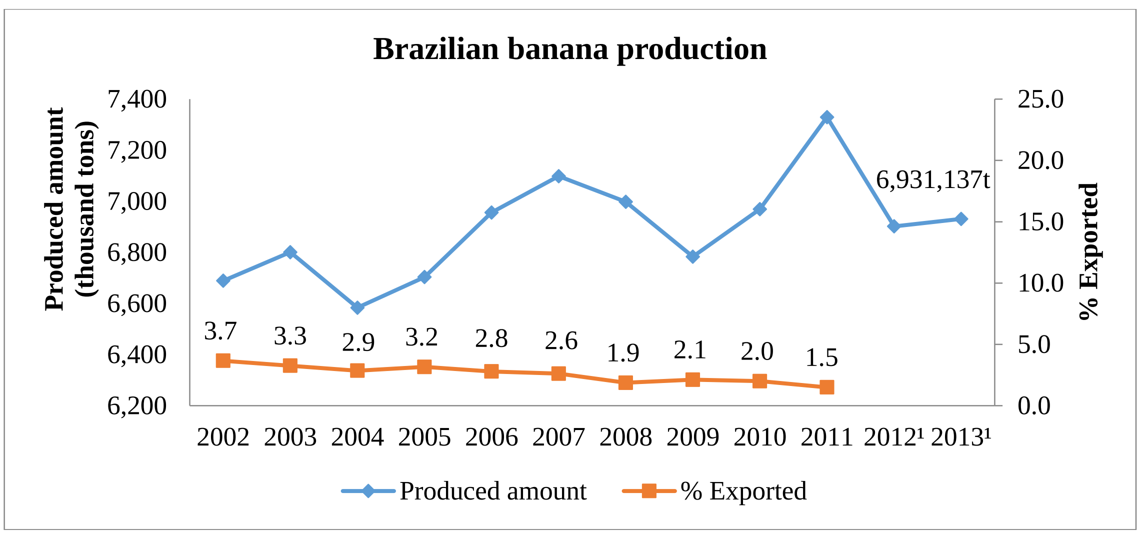

Figure.1. Brazilian banana production historical series (IBGE, 2014a, 2014b) and exports (SECEX, 2014).

Another reason for the low level of postharvest care in Brazil is that the largest part of domestic production comprises bananas of the *Prata* subgroup, which are more resistant to inadequate handling in the postharvest period (Lichtemberg, Lichtemberg, 2011). According to Reinhard et al. (2013), only 2% to 5% of Brazilian banana production is exported, domestic consumption amounts between 65% and 70% of production and postharvest losses are approx. 30%. Banana productivity in Brazil, between 13 and 14.5 t/ha (FAO, 2014), is still low when compared to the global average of approx. 20 t/ha. Europe has a rate of 36 t/ha and Asia 28 t/ha (EMBRAPA, 2009).

The main banana producing regions in Brazil are the Northeast (34.1%) and Southeast (33.5%), as shown in Figure 2. The value of production for these two regions, in 2012, was 3,162,691 thousand Brazilian reais (IBGE, 2014b).

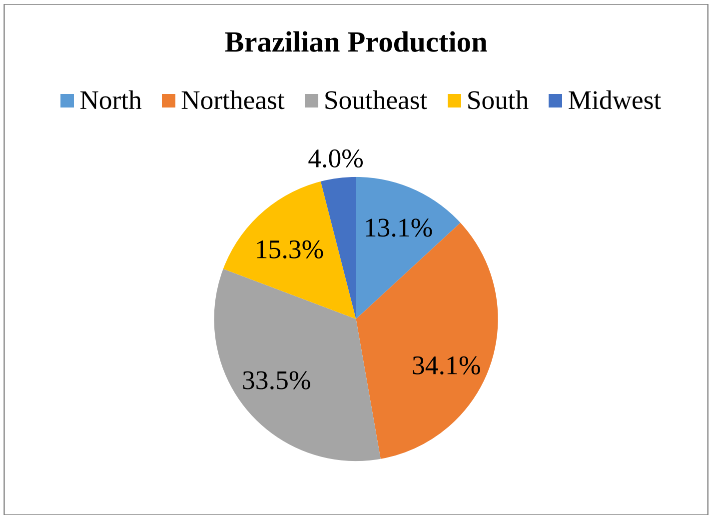

Figure 2. Regional production share for the 2013 banana crop (IBGE, 2014a).

The main banana producing states in Brazil are: São Paulo, Bahia, Santa Catarina, Minas Gerais and Pará, together accounting for 61% of domestic banana production and over 63% of the value of the 2012 crop, as shown in Table 2 (IBGE, 2014b).

Table 2. States with the highest share of domestic banana production in 2012 (IBGE, 2014b).

State	Yield (t)	Share of Domestic Production (%)	Value of Production (000´s Brazilian Reais)	Share of the value of Domestic Production (%)	Productivity (t/ha)
São Paulo (SP)	1,215,435	17.6	851,210	19.36	22.5
Bahia (BA)	1,083,346	15.7	734,725	16.71	15.0
Santa Catarina (SC)	689,815	10.0	275,528	6.27	23.3
Minas Gerais (MG)	687,293	10.0	639,030	14.54	16.4
Pará (PA)	547,098	7.9	275,409	6.26	13.1
Ceará (CE)	415,763	6.0	217,275	4.94	8.8
Pernambuco (PE)	407,574	5.9	168,451	3.83	9.7
Paraná (PR)	276,890	4.0	127,579	2.90	24.0
Espírito Santo (ES)	241,997	3.5	151,224	3.44	11.3
Goiás (GO)	197,990	2.9	155,626	3.54	15.8
Brazil (Total)	6,902,184	100.0	4,396,349	100.00	14.1

In 2012, the municipalities where the main banana producers were located are the following: Corupá/SC (2.4% of domestic production), Miracatu/SP (2.3%), Cajati/SP (2.1%), Bom Jesus da Lapa/BA (1.8%) and Luiz Alves/SC (1.8%). In terms of the value of production, the following are worthy of mention: Cajati/SP, Sete Barras/SP, Eldorado/SP, Wenceslau Guimarães/BA and Jaíba/MG. As can be seen in Table 3, the top ten municipalities for domestic banana production account for less than 20% of the share of domestic production as well as value of production. This demonstrates the wide geographical distribution of banana producers throughout the country (present in the 27 states of Brazil as well as the federal district of Brasilia), with only a few centers that stand out as the largest producers, as shown in Figure 3. In 2012, the vast majority of producing municipalities had a turnover of as much as 3 million Brazilian reais. Only Cajati/SP and Sete Barras/SP earned more than 90 million Brazilian reais from banana production. The strategic banana producing regions are the follow: the Southern coastal region of the state of São Paulo, the North of Minas Gerais, the Southern coastal region of the state of Bahia, the Southwest of Pará and the Northern Santa Catarina.

Table 3. Municipalities with highest values of banana production in 2012 (IBGE, 2014b).

Municipalities	Quantity produced (t)	Share of Domestic Production (%)	Municipalities	Value of production (´000 Reais)	Share of value production (%)
Corupá - SC	165,420	2.4	**Cajati - SP**	130,950	3.0
Miracatu - SP	158,400	2.3	**Sete Barras - SP**	100,000	2.3
Cajati - SP	145,500	2.1	**Eldorado - SP**	87,750	2.0
Bom Jesus da Lapa - BA	127,179	1.8	**Wenceslau Guimarães - BA**	83,680	1.9
Luiz Alves - SC	127,100	1.8	**Jaíba - MG**	82,820	1.9
Wenceslau Guimarães - BA	115,900	1.7	**Jacupiranga - SP**	70,200	1.6
Sete Barras - SP	100,000	1.4	**Registro - SP**	69,750	1.6
Eldorado - SP	97,500	1.4	**Bom Jesus da Lapa - BA**	69,313	1.6
Guaratuba - PR	96,480	1.4	**Luiz Alves - SC**	68,507	1.56
Jaíba - MG	82,000	1.2	**Janaúba - MG**	58,888	1.34
Total	1,215,479	17.6	**Total**	821,858	18.7

This study was concentrated on several centers that stand out as being the largest producers, namely Northern Santa Catarina, Vale do Ribeira, Northern Minas Gerais, Bom Jesus da Lapa - BA, Vale do Submédio do São Francisco, Vale do Açu - RN and Vale do Jaguaribe - CE and the Southern and Central-Southern regions of the state of Bahia. Figure 3 locates these production centers and the arrows indicate the main destination markets. Production in these centers is, for the most part, intended for the large cities in the region, however the states with the highest exports are Rio Grande do Norte, exported 21.8% of its production to Europe in 2011 (31,000 tons, corresponding to 28.3% of exports from Brazil); Santa Catarina, which exported 8.3% of its production (54,300 tons, or 49.3% of Brazilian exports) to MERCOSUR countries (Uruguay and Argentina); and Ceará, which exported 23,000 tons of bananas to Europe (21% of Brazilian exports) (SECEX, 2014, IBGE, 2014b).

The distances considered for the assessment of the environmental impact of bananas transporting in Brazil, as well as the burdens associated with this life cycle stage are shown in Table 4. Figure 4 shows the contribution of the different scenarios to the burdens evaluated.

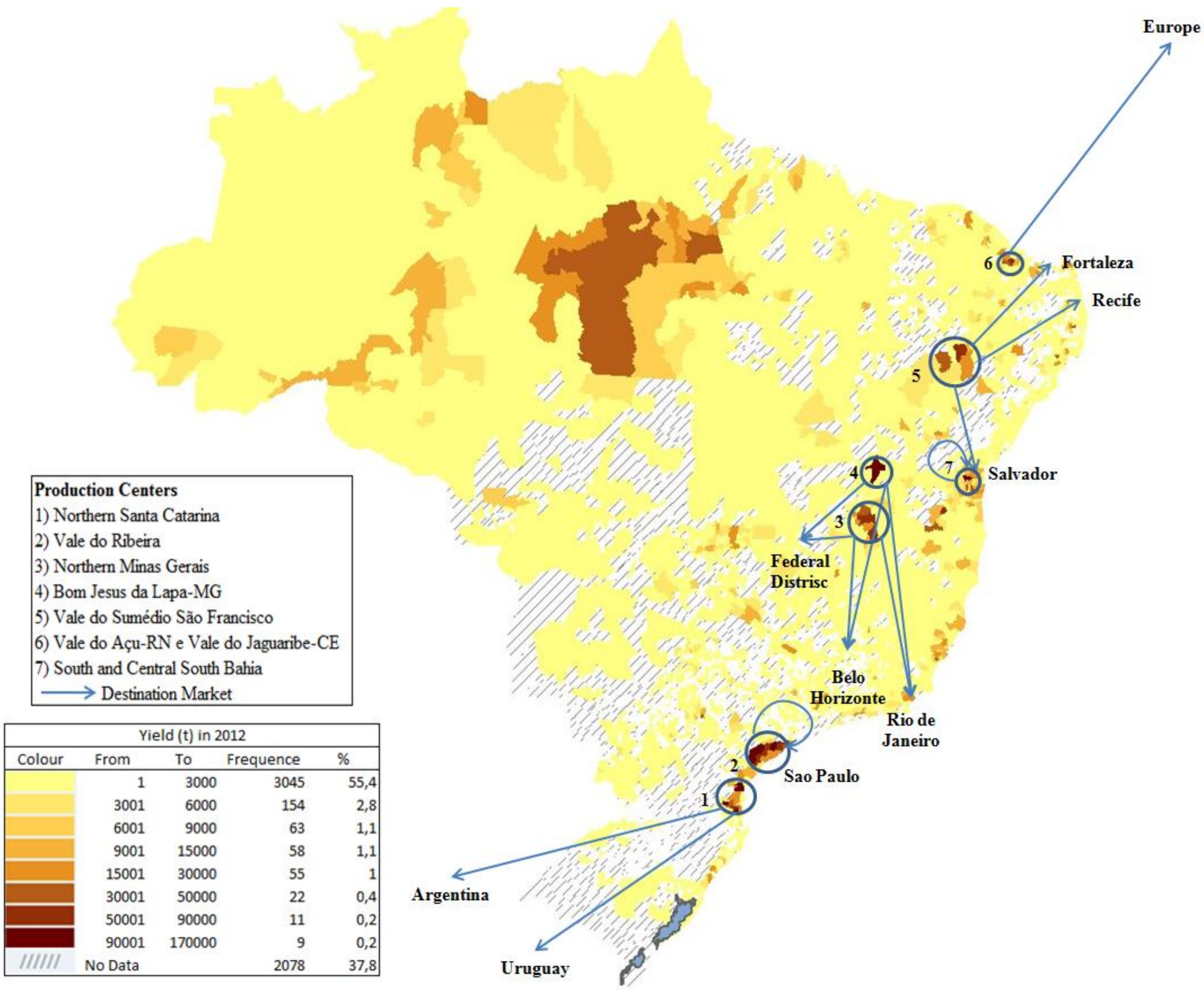

Figure 3. Banana production hotspots in Brazil and main destination market (IBGE, 2014b; Embrapa, 2009).

Table 4. Burdens associated with the transport of bananas in Brazil (Functional unity = 1 ton).[1]

Production center / Distance from domestic markets[2]	Burdens	Scenario 1 Return full	Scenario 2 Return empty	Scenario 3 Return full and 33% losses in T
Northern Santa Catarina State / 378 km	CO_2 eq (kg)	22.72	45.44	33.86
	Energy use (MJ)	49.06	98.13	73.11
	Non renewable resources (kg)	8.24	16.48	12.28
Vale do Ribeira / 193 km	CO_2 eq (kg)	11.60	23.20	17.29
	Energy use (MJ)	25.05	50.10	37.33
	Non renewable resources (kg)	4.21	8.41	6.27
Northern Minas Gerais State / 909.7 km	CO_2 eq (kg)	54.68	109.37	81.48
	Energy use (MJ)	118.08	236.16	175.94
	Non renewable resources (kg)	19.83	39.65	29.54
Bom Jesus da Lapa - BA / 1207.85 km	CO_2 eq (kg)	72.61	145.21	108.18
	Energy use (MJ)	156.78	313.55	233.60
	Non renewable resources (kg)	26.32	52.65	39.22
Vale do Submédio São Francisco / 717.5 km	CO_2 eq (kg)	43.13	86.26	64.26
	Energy use (MJ)	93.13	186.26	138.76
	Non renewable resources (kg)	15.64	31.27	23.30

Table 4. Burdens associated with the transport of bananas in Brazil (Functional unity = 1 ton).[1] (continuation)

Vale do Açu - RN and Vale do Jaguaribe - CE / 281.5 km	CO2 eq (kg)	16.92	33.84	25.21
	Energy use (MJ)	36.54	73.08	54.44
	Non renewable resources (kg)	6.14	12.27	9.14
South and Central South Bahia State / 277.7 km	CO2 eq (kg)	16.69	33.39	24.87
	Energy use (MJ)	36.04	72.09	53.71
	Non renewable resources (kg)	6.05	12.10	9.02
Exports from Northern Santa Catarina State to Argentina and Uruguay / 3,009.7 km	CO2 eq (kg)	211.55	423.10	315.21
	Energy use (MJ)	456.79	913.58	680.62
	Non renewable resources (kg)	76.70	153.40	114.28
Exports from Vale do Açu - RN and Vale do Jaguaribe - CE to Europe (mainly Germany) / 9,000 km	CO2 eq (kg)	149.11	298.21	222.17
	Energy use (MJ)	276.18	552.36	411.51
	Non renewable resources (kg)	46.37	92.75	69.10

[1] Transport with 80% load in all scenarios
[2] Average Distance Covered (one-way) in the Domestic Market.
[3] Average Distance Covered (one-way) for Exports, to the receiving port.

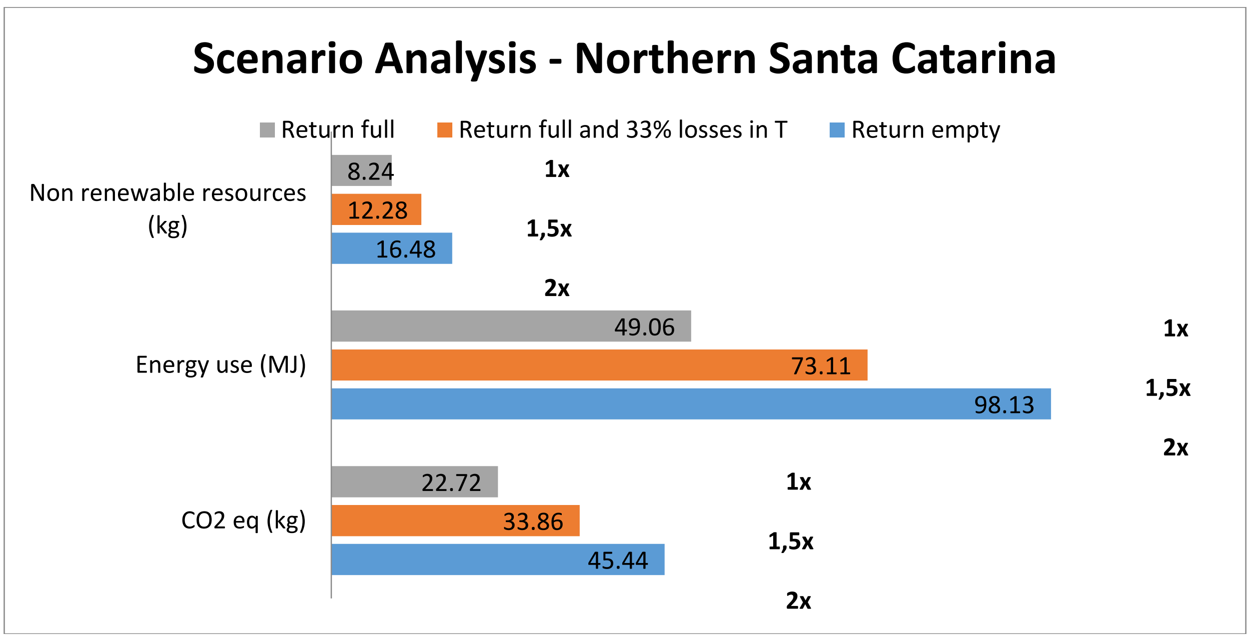

Figure 4. Contribution of the different scenarios to the burdens evaluated.

4. Discussion

Brazil has seven banana producer regions distributed along the country. Most of the banana produced in Brazil are consumed near the production centers, except for Northern Minas Gerais and Bom Jesus da Lapa - BA regions which distances from the consumers are approx. 1,000 km.

The banana consumed in São Paulo has the lowest environmental impact related to the transport stage since it was produced in the Vale do Ribeira region that is the region with the shortest distance between production site and consumers (193 km).

On the other hand, the banana consumed in Rio de Janeiro, Brasília and Belo Horizonte has the highest environmental impact related to the transport stage since it is produced in the Bom Jesus da Lapa - BA region which is the region with the highest distance from the consumers (approx. 1200 km).

There have been various studies on the LCA of bananas, the majority having a "cradle-to-gate" scope (from production to retail in Europe), except for Kilian et al. (2012) who took into account the “cradle-to-grave” scope which includes up to consumption stage and the study by Svanes & Aronsson (2013), which took both scopes into consideration. Kilian et al. (2012) stated that this tropical fruit emits 1.09 kg of CO_2-eq per kg of exported bananas, with overseas transport being responsible for 78% of emissions, followed by agricultural production with 15% and distribution with 7%. Lescot (2012) calculated that the carbon footprint of bananas exported to Europe is 0.85 kg of CO_2e per kg of bananas, the stages with the greatest impact being overseas transport (43%), agricultural production (29%) and packing (12%). Luske (2010) estimated the carbon footprint at 1.12 kg of CO_2-eq per kg of exported bananas and also showed overseas transport and agricultural production to be the stages with the greatest impact, whose contributions were 62% and 12%, respectively. Svanes and Aronsson (2013) found the carbon footprint (from farm-to-retail) of 1.37 kg of CO_2-eq/kg and concluded that the banana hot spots in their study were overseas transport, which accounted for around 55% of the carbon footprint, followed by agricultural production, which accounted for 16%.

Differently from the other studies on LCA of banana available in the literature, the export stage of Brazilian banana to Europe is the life cycle stage with the lowest environmental impact due to the optimum relation of the load per trip ratio of the ships which minimizes the consumption of fuel per ton of transported banana.

Surprisingly despite a travelling distance 3 times larger from the northeastern Brazil to Europe than Santa Catarina to MERCOSUR the average overseas transport impacts are 4.2 to 4.5 times smaller (436.13%) than export via truck to Argentina and Uruguay. Besides, the exports of banana to Mercosur is the highest environmental impacting stage since it is based on truck transportation while all the transportation to Europe are made by ship.

Apart from the higher consumption of fuel per ton of banana transported by trucks than ships, the distances of the producer regions from domestic market can reach values over 1,000 km due to the continental dimensions of Brazil which contributes to enhance the environmental impact of this stage of the life cycle.

The data shows that in the domestic market the burdens vary in more than 80%, that is, the banana consumed in São Paulo has an estimated transport emission of 23.2 kg of CO_2 eq, 50.1 MJ of energy use and 8.41 kg of non renewable resources, while the largest environmental impact of the banana that is produced in Bom Jesus da Lapa – BA, serving cities in Rio de Janeiro, Brasília and Belo Horizonte has emissions of 145.21 kg of CO_2 eq., 313.55 MJ of energy use and 52.65 kg of non renewable resources. This variation is smaller when analyzing exports to MERCOSUR and Europe: (29.5%) in CO_2 eq, and 39.5% of energy use and non renewable resources.

Taking into account the different scenarios, the impact of the transport of bananas in scenario 3 (truck returns full and 33% banana losses in transport) is 1.5 times higher than scenario 1 (truck returns full). If the cultivation stage is being accounted, the impact of banana loss is even higher. So, efforts must be made to reduce the product loss through the life cycle.

5. Conclusion

The results have shown that the banana consumed in São Paulo has the lowest environmental impact related to the transport stage since it was produced in the region with the shortest distance between production site and consumers.

On the other hand, the banana consumed in Rio de Janeiro, Brasília and Belo Horizonte has the highest environmental impact related to the transport stage since it was produced in the region with the highest distance from the consumers.

Despite three times higher distance, the exports to Europe has 70% lower environmental impact than exports to MERCOSUR since it is based on ship transportation while transportation to Argentina and Uruguay is made by truck.

This study reinforces the importance of the transport modals and reduction of food loss in reducing the product environmental impacts.

Acknowledgements

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Design for environment applied to rice production

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ABSTRACT

Life cycle thinking was applied to the rice production system in order to evaluate the possibilities of environmental impact reduction. Improvements have been applied to several stages of the life cycle as follow: 1) Irrigation, 2) Drying, 3) Core and clean stage, 4) Packaging and 5) Transport. The work was carried out from June 2012 to August 2013. The functional unity adopted was 1,000 kg of packed rice (5 kg packs), available at retail. The system boundary considered the field operation, including transportation after harvest, fertilizer production, packaging and transport until the retail. The results have shown the new rice production system has significant environmental performance improvement, mainly water (approx. 200,000 L/t) and energy consumption (approx.. 200 MJ/t) as well as GHG emission (approx.. 30 kg CO2 eq/t) and yield (1,152 kg/ha), being the new irrigation system responsible for the majority of the benefits.

Keywords: rice, irrigation, environmental performance, design for environment, life cycle thinking

1. Introduction

Rice cropping fields are important contributor to climate change since they are a major methane source. The area of rice cropping fields in the world is gradually increasing and this area is expected to continue expanding as the world population increases. So, mitigating methane emission in these areas is an important issue. This can be accomplished through water management practices. According to study developed in central Japan by Kudo et al (2014), a compound treatment with a combination of flooding, midseason drainage and intermittent drainage may be an effective water management practice for mitigating greenhouse gas emission and maintaining rice yield.

Study developed by Perret et al (2013) on rice production in the North-eastern Thailand in 2010 evaluated 45 diverse rice cropping systems according to three systems, namely wet-season rain-fed, wet-season irrigation, and dry-season irrigation systems. According to the authors, a wide-ranging performances and impacts were observed, despite cropping practices were relatively homogeneous. The differences among the systems were originated mostly from differences in yield, which were largely impacted by water supply. The results showed the low performances and high impacts of dry-season irrigated systems, since they require mostly blue water, while the two other systems rely primarily on green water. Besides, the dry-season irrigated systems require more energy, labor, fertilizers, pesticides, and ultimately yield lower production.

Khoshnevisan et al (2014) applied LCA to evaluate the environmental impact of consolidated rice farms (CF) – farms that have been integrated to increase the mechanization index, and traditional farms (TF) – small farms with lower mechanization index, in Guilan Province, Iran. The results have shown that the energy ratios for CFs and TFs were 1.6 and 0.9, respectively. The two main reasons for the higher energy ratio by the CFs are the total input energy in the CFs was less than that of the TFs and the CFs employed better agricultural management, including the use of higher-yielding varieties of rice that generated higher yields. Then, the results showed that CFs produced fewer environmental burdens per ton of produced rice. The same conclusion was reached for land-based functional unity, which indicates that the differences between the two types of farms were not caused by a difference in their production level, but rather by improved management on the CFs. The study also showed that electricity accounted for the greatest share of the impact for both types of farms, followed by P-based and N-based chemical fertilizers. The LCA results indicated that the use of more mechanical power and an associated increase in the mechanization index will lead to better environmental performance rather than increased environmental burdens from rice cultivation. Besides, the cultivation of higher yielding rice varieties and the use of agricultural inputs according to crop requirements and soil analyses can improve the environmental performance as well.

Rice straw can cause environmental impacts if burned in situ in the field. Then, Silatertruksa and Gheewala (2013) conducted life cycle assessment of four rice straw utilization systems, which are (1) direct combustion for electricity, (2) biochemical conversion to bio-ethanol and biogas, (3) thermo-chemical conversion to bio-DME, and (4) incorporation into the soil as fertilizer, in order to compare their environmental performances. According to the authors, the bio-ethanol pathway resulted in the highest environmental performance in relation to reductions in global warming and resource depletion potentials. Rice straw bio-DME was preferable in relation to reduction in acidification potential. Rice straw electricity and fertilizer brought several environmental benefits. However, removal of agricultural residues must be managed carefully since excessive residue removal can degrade the long-term productive capacity of soil resources.

Rice (160.3 g/day) and beans (182.9 g/day) are the basis of Brazilians food consumption (IBGE, 2011), that is why rice has been chosen as one of the 18 products of the project “End-to-End Sustainability” developed by Walmart Brasil and its main suppliers (Walmart, 2013). In this project, Pilecco Nobre Alimentos Ltda. implemented several improvements in the rice production with the aim of reducing the environmental impact of the product with the assistance of CETEA/ITAL. Life cycle thinking was applied to the rice production system in order to evaluate of possibilities of reducing the environmental impact of the rice production.

2. Methods

This work was conducted in accordance with the International Standard ISO/TR 14062 (ISO/TR 14062, 2002). Taking into account the life cycle thinking, improvements have been applied to several stages of the life cycle of rice production as shown in Table 1. The goal of this work was to reduce the environmental impact of rice production. Traditional rice production system was established as a baseline case against which the impacts of the conditions could be quantified.

Table 1. Life cycle stages and respective improvements applied to the rice production.

Life cycle stage	Traditional production system (baseline)	New production system
Irrigation	Irrigated rice cropping system	Underground drip irrigation rice cropping system
Drying	Firewood furnace and temperature rate system with high grains breakage	Steam radiators and homogeneous temperature system with reduced grains breakage
Core and clean	Without rice recovery	Rice recovery after straw removal
Packaging	Plastic packaging from non-renewable resource	Plastic packaging from renewable resource, sugar cane
Transport	Conventional diesel truck fleet	Diesel S10 and more efficient truck fleet

The work was carried out from June 2012 to August 2013. The new irrigation system was implemented by Pilecco Nobre Alimentos Ltda. in a cultivation area of 64 hectares, in Alegrete, Rio Grande do Sul, Brazil.

The functional unity adopted was 1,000 kg of packed rice (5 kg packs), available at retail. The system boundary considered the field operation, including transportation after harvest, fertilizer production, packaging and transport until the retail (Figure 1). Only the steps that have been improved were taken into account. The emissions from fertilization and methane emission from rice fields were calculated using IPCC models (IPCC, 2006).

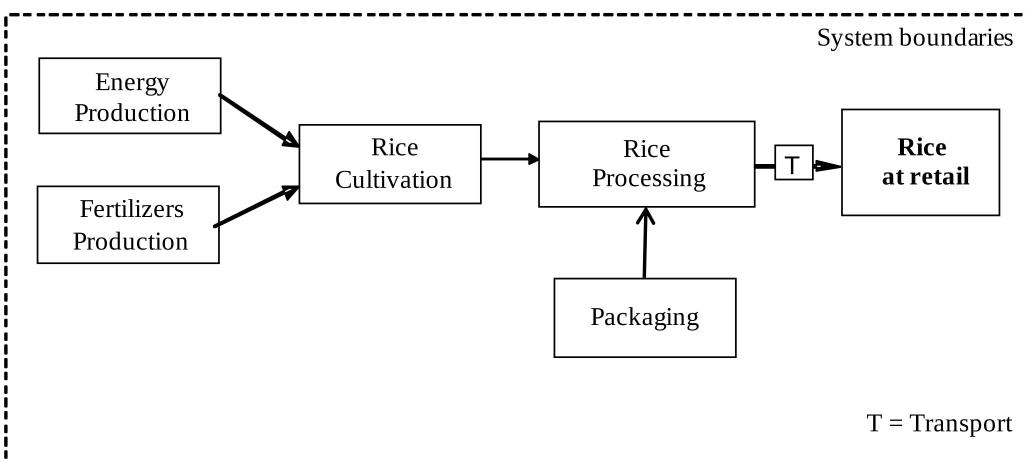

Figure 1. System boundaries adopted in this work.

Farm specific data along with industrial production data have been combined in order to model a rice production system. GHG emission factors for transport and energy production were obtained from GHG Protocol Brasil (2012). The environmental aspects relative to the fertilizers production were taken from recognized database and included in the boundary.

The inventory quantities fossil/renewable primary energy demand (PED), land use and water use were analyzed. GWP (100 years), eutrophication (EU) and acidification (AP) were estimated according to the CML method (GUINÉE, 2002).

3. Results

The improvements applied to rice cultivation, processing and transport are described below as well as the gains of these changes to the environmental impact of the product.

3.1. Irrigation

The rice cropping system used in the traditional production system – baseline case (Figure 2a) employs continuous irrigated field, which needs a pumping system with a high flow demand since this process is based on water drainage from the pumping point to all the crop area. Due to the high extension of the irrigated area, this takes several days to complete irrigation. Besides, in this process occurs a great water loss due to evaporation and shifting in the channels, as well as contamination of the catchment rivers by fertilizers and pesticides.

(a) (b)

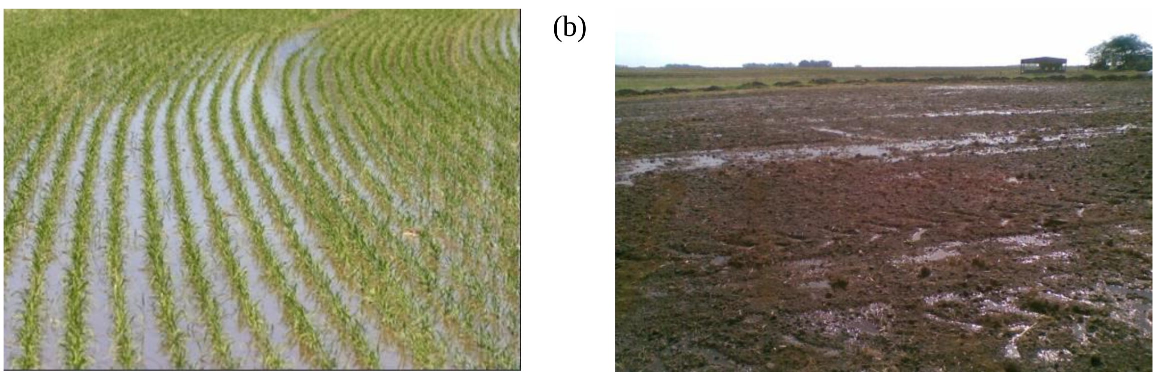

Figure 2. Irrigation of rice cropping system moved from: (a) Irrigated system to (b) Underground drip irrigation system.

The substitution of the irrigated system by the underground cropping system (Figure 2b) allows absorption of the water directly by the radicular system of the plant avoiding water loss by evaporation and shifting since it is based on a system of channels that directs the water flow always to the same place. Besides, the underground cropping system allows all fertilizers and pesticides be applied without loss.

The installed pumping power of the underground drip irrigation system is significantly lower than the irrigated system due to the higher water flow in the canalizations. As a result, the underground drip irrigation system has lower energy and water use than the irrigated system, besides lower aquatic eutrophication and ecotoxicity.

3.2. Drying

The rice drying process consists of reducing the humidity of the rice grain, still with straw, to values of approx. 10% in order to avoid fermentation and pest propagation (worm, moth, caterpillar). The traditional processes use dryers with furnaces feed by rice straw or firewood to generate hot gases, which gradually remove the humidity from the rice grain.

If this process is not well controlled it can degrade the grain by breakage due to formation of temperature gradient. The breakage of the rice grain can also occurs due to the high recirculation the traditional dryers require in order to get high productivity. Broken grains are not used as quality rice.

So, in the second semester of 2012 a drying system that uses vapor from thermal power plant feed by rice straw instead of firewood was installed - GEEA project - Alegrete Electric Energy Producer Ltd.. Besides elimination of the firewood, this new system has a differentiated system of drying rack that allows a more homogeneous distribution of the hot air avoiding the breakage of the grains due to temperature gradient, as shown in Figure 3. Since this drying system is more efficient than the traditional one, it reduces the recirculation of grains inside the equipment, which decreases the breakage by moving.

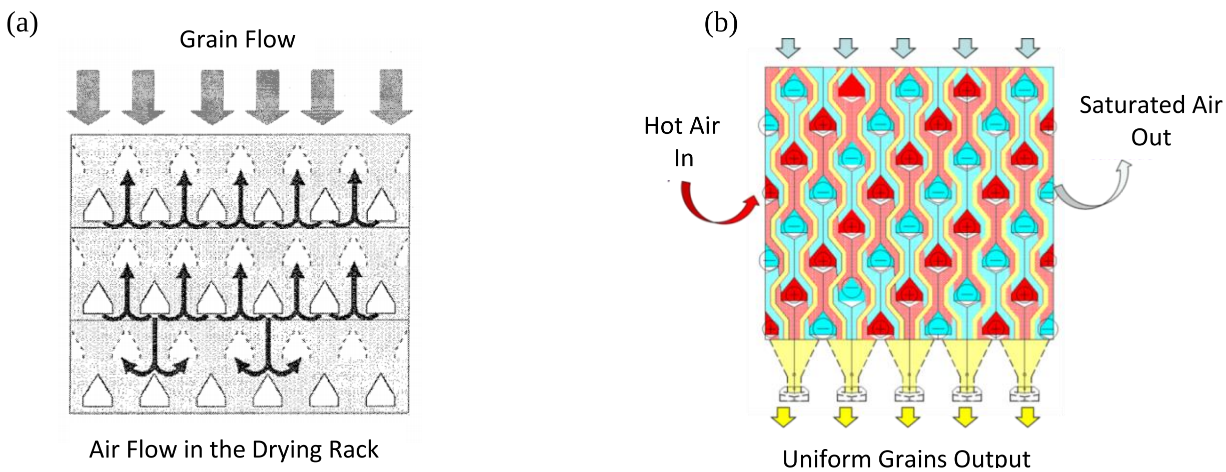

Figure 3. Drying systems: (a) traditional drying rack, (b) new system of drying rack.

The benefits of this improvement are the use of renewable energy from waste as well as a better yield of whole rice grains.

3.3. Core and clean

All rice straw generated by Pilecco Nobre is sent to GEEA - Alegrete Electric Energy Producer Ltd., where it is used as raw material to produce electric energy and silica. However, a fraction of rice without straw as well as rice still with straw is lost during the straw removal process, being forwarded with the straw itself to electric energy generation.

The new rice production system has a rice recovery from straw (Figure 4) which works by weight difference between the straw and the grain. This improvement allowed to recover the rice that used to be burnt with the straw and then increased the yield of the productive system.

Figure 4. Equipment for rice recovery after straw removal.

3.4. Packaging

The packaging of the rice was changed from polyethylene from non-renewable resource to polyethylene from renewable resource (ethanol from sugar cane). The polyethylene from renewable resource has the same properties, performance and versatility of applications as the polyethylene from non-renewable resource, which makes easy its use. For the same reason it is recyclable in the same recycling chain of the traditional polyethylene.

The benefits of this improvement are the use of renewable resources and reduction of non-renewable resources consumption (oil), as well as decrease of GHG emissions.

3.5. Transport

The transport of rice used to be in a truck fleet based on traditional diesel consumption, which emits 500 mg of sulfur per liter of diesel and has an average fuel consumption of 0.33 L of diesel per km. The improvement of this life cycle stage was based on the substitution of seven trucks by new trucks that uses diesel S10 which emits 10 mg of sulfur per liter of diesel, besides an average fuel consumption of 0.29 L of diesel per km.

The benefits of this improvement are reduction of non-renewable resources consumption (oil), as well as lower GHG emissions and acidification potential than the previous fleet.

Table 2 summarizes the relative reduction of the environmental impacts considered in this work, taking into account all the improvements made in the rice production system, while Figure 5 shows the contribution of each improvement.

Table 2. Improvement of the environmental performance of the rice production system evaluated: Traditional vs New production system. (Functional unity = 1,000 kg of packed rice).

Parameter	**Improvement**
Yield (kg/ha)	1,152.36
Primary energy demand (MJ)	-234.85
Electric energy consumption (MJ)	-206.94
Non-renewable energy (MJ)	-26.45
Water consumption (L)	-198,505.31
Non-renewable resources (kg)	-7.61
Firewood consumption (kg)	-71.55
Rice waste (kg)	-1.42
Land use (ha)	-0.03
GWP (100yr) (kg CO_2 eq)	-31.43
Aquatic eutrophication (kg PO_4---eq)	-15.75
Aquatic acidification (kg SO_2 eq)	-46.16

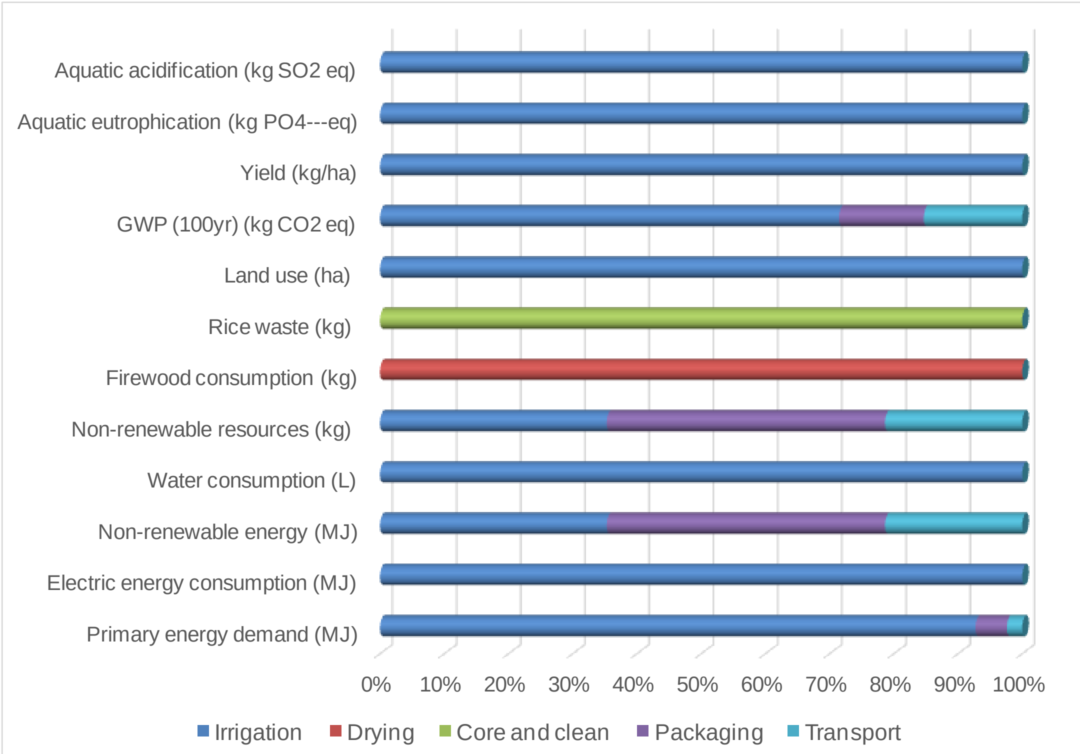

Figure 5. Contribution analysis of the improvements made in the rice production system to the impact categories evaluated.

4. Discussion

The combined modifications clearly produced substantial benefits for the environmental performance of rice production, mainly reduction of water, energy and firewood consumption as well as GWP and aquatic acidification.

The key opportunity in terms of environmental benefit is the reduction of water consumption (approx. 200,000 L/t of rice) which is quite relevant due to the water scarcity we are facing nowadays because of the climate change. According to the last IPCC report (2014), the already observed and the projections of climate change for the South America will compromise the water availability for this region and direct impacts should occur in the domestic and industrial water supply as well as in sectors strongly water dependent as hydroelectricity production and agriculture.

Besides, the yield increased by 15%, which means less land use for cultivation, reduction of inputs (fertilizers, pesticides etc.) and associated emissions, besides feeding more people.

The new irrigation system was responsible for the majority of the environmental impact reductions, i.e. electric energy and water consumption, land use, yield, aquatic eutrophication and acidification. Besides, this improvement contributed also to reduce the non-renewable resources consumption and GHG emissions.

The new drying system contributed to the reduction of firewood consumption, while the improvement made in the core and clean stage reduced the rice waste.

The change in the packaging material contributed to reduce the non-renewable resources and non-renewable energy use as well as GHG emissions.

The modification made in the transport stage contributed to reduce the same burdens than the new packaging, but with lower reduction of the non-renewable resources and non-renewable energy use.

5. Conclusion

This work supplied important results for the better environmental performance of the rice production, making this product a lower environmental impact than the traditionally cultivated rice.

An expressive reduction of water consumption was observed in the rice cultivation due to the substitution of the irrigated rice cropping system by the underground drip irrigation rice cropping system, besides other benefits like reduction of energy use, fertilizer and pesticide loss and GHG emissions.

The 15% increased yield was mainly a result of the new cultivation system.

Acknowledgement

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Energy risk management as a driver for reducing greenhouse gas emissions from farming

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ABSTRACT

Instead of focusing on GHG emissions, farmers might be more willing to implement improvements in farming systems if the benefits were shown in terms of energy use and energy risks. Reducing energy costs has a direct benefit to farmers, and moving a farm's net energy budget towards a surplus improves the farm's resilience for energy price fluctuations. We developed a tool for strategic energy management analysis of farms. The tool uses life cycle assessment approach for estimating the embodied energy inputs of the farm, and quantifies the energy outputs in form of agricultural products and surplus of renewable energy produced at the farm. We demonstrate the benefits of the tool by using a case study of a dairy farm in the United Kingdom. We compare the energy balance and GHG emissions of the farm with two alternative scenarios. The results show that synergies between energy risk management and GHG mitigation exist.

Keywords: agriculture, embodied energy, greenhouse gas emissions, tools, energy risk

1. Introduction

Many tools for estimating greenhouse gas (GHG) emissions from farming are widely available (Bochu et al. 2013; Hillier et al. 2011). However, the farmers' willingness to use such carbon calculators is often low due to lack of direct benefits gained by using them (Elbersen et al. 2013). Instead of focusing on GHG emissions, farmers might be more willing to implement improvements in their farming systems if the benefits were shown in terms of energy use and energy risks. Reducing energy costs has a direct benefit to farmers, and moving a farm's net energy budget towards a surplus improves the farm's resilience for energy price fluctuations. Our hypothesis is that reduction in embodied energy inputs and increase in on-farm bioenergy production helps to reduce GHG emissions, partly because of synergies between energy, but largely because it motivates farmers to take action.

We developed a tool, called Energy Positive, for strategic energy management analysis of farms. The tool uses life cycle assessment approach for estimating the embodied energy inputs of the farm, and quantifies the energy outputs in form of agricultural products and surplus of renewable energy produced at the farm. The assessment helps to identify the main energy inputs of the farm and to compare alternative farm management scenarios in terms of energy risks. The tool shows how the farm's input costs reflect fluctuations in energy prices.

We demonstrate the benefits of the tool by using a case study of a 350 ha dairy farm located in the United Kingdom. We compare the current energy balance of the farm with two alternative scenarios: a low GHG emission scenario and a high renewable energy production scenario. In order to test our hypothesis, we analyze the GHG emissions of those scenarios by using the Carbon Calculator developed by the European Commission (Bochu et al. 2013).

2. Methods

2.1. General approach

LCA was used for comparing energy balances and GHG balances of a case study dairy farm located in the Southern England, in the United Kingdom. The impacts of various management choices on GHG emissions and energy balance were modelled. The functional unit (FU) was the whole farm.

The system boundaries included the production of farming inputs (e.g. fuels, fertilizers and pesticides), machinery manufacturing and farm operations, including crop cooling and drying. The soil carbon emissions and sequestration were not taken into account, because net sequestration or emission only occurs when the soil management type has been changed until a new equilibrium level is reached. Energy balances were calculated by using the Energy Positive tool, whereas GHG emissions were calculated with European Commission carbon calculator (available from: http://mars.jrc.ec.europa.eu/mars/Projects/LC-Farming). The methodology of the EC's carbon calculator has been explained elsewhere (Bochu et al. 2013). The Energy Positive tool calculates the LCA based primary energy balances of farms based on data from Williams *et al.* (2006). The biogas yields for manure and straw were based on data from Michel et al. (2010)

2.2. Case study farm

The case study farm is a 350 ha dairy farm located in southern England in the United Kingdom. The farm has 200 milking cows (production level 6000kg milk/cow/yr), 50 heifers and 50 calves. All feed is produced at the farm utilizing 253 ha (Table 1). The livestock is grazing 50% of the time in a year.

Table 1. Details of the feed production system.

Area (ha)	**Crop**	**Tillage**	**N fertilizer (kg/ha)**	**Manure (kgN/ha)**	**Pesticides (yes/no)**	**% clover in grassland**	**Yield (t/ha)**
25	Corn silage	full	200	50	Herb, fung		40
25	Wheat	low	150	50	Herb, fung		7
60	Clover-grass	full	30	no	Herb, fung	50	11
10	Barley	full	120	25	Herb, fung		5
133	Permanent pasture	No-till	no	no	no	no	7

2.3. Alternative production scenarios

The impacts of some alternative production scenarios were assessed. To enable smooth comparison, the farm area, number of animals and product output were assumed to be the same in each scenario. The impacts of the following practices were assessed:

1) No-tillage: applied to the whole agricultural area.
2) Reduced use of synthetic nitrogen fertilizer: the use of synthetic N fertilizer was halved for corn silage and wheat. Barley was replaced with a legume crop. The clover content of the clover-grass was increased to 75%.
3) Anaerobic digester: the livestock manure and straw from wheat was assumed to be used for biogas production. It was assumed that 50% of the manure produced by the whole cattle was used for anaerobic digester, resulting in biogas yield of 50,000 m^3 (22.2 MJ/m^3). The biogas yield for straw was assumed to be 7.1 GJ tDM^{-1} (Berglund and Börjesson 2006), and straw yield 3tDM/ha. It was assumed that 12% of the biogas energy was used for the running the biogas reactor (heating, pumping and mixing). The biogas was assumed to be used for electricity generation, yielding 30% electricity and 50% heat. The rest was assumed to be lost.
4) Solar panels: were assumed to be installed on south-facing roof surface area (500 m^2).

2.4. Method for estimating GHG reduction potential of the mitigation actions

The methods for estimating the GHG mitigation potential of the mitigation actions are explained in more detail in (Bochu et al. 2013) and only summarized here:

1) No tillage: the mitigation effect is calculated based on increase in soil carbon, reduction of CO2 emissions from fuel consumption and increased N2O emissions.
2) Reduced use of synthetic nitrogen fertilizer: avoided emissions from manufacturing of synthetic N fertilizers, and the change in emissions due to replacement of barley with a legume crop and increase of

the clover content in clover-grass. The emission factor for synthetic N fertilizer production is based on data from Wood and Cowie (2004), varying between 5.1-7.1 t CO_2-eq/t N depending on the type of N fertilizer used.

3) Anaerobic digester: avoided N_2O and CH_4 emissions from manure storage, and avoided emissions from manufacturing of N fertilizers due to reduced N losses as NH_3 and N_2O.
4) Solar panels: avoided emissions from electricity production (average UK electricity mix used as a reference).

3. Result

Table 2 shows the GHG mitigation actions recommended by the carbon calculator and the mitigation potential per ha for the base scenario. The mitigation potential for no-tillage and biogas production are different between Table 2 and 3 due to the fact that the mitigation potential presented in Table 2 includes increase in carbon storage in the soil whereas that is not included in the results presented in Table 3.

The results in Table 3 show that in the case of milk production, there is not direct correlation between energy input and GHG emissions. The main GHG emissions sources are enteric fermentation, N_2O emissions from soils and manure management, which all are independent of energy use. The main energy inputs are electricity use and production of fertilizers.

The results also show that combined effect of all mitigation actions included in the study resulted in nearly energy positive farm even when not considering the surplus energy produced at the farm. The same actions however, reduced only 23% of the GHG emissions.

Table 2. Results of mitigation action recommendations generated by the carbon calculator for the base scenario.

Rank	**Actions**	**tCO_2e saving/ ha/ year**	**New level of tCO_2e/ ha/ year**	**% saving**
	Current situation		5.34	
1	No-tillage	0.50	4.85	9.3%
2	Biogas production	0.41	4.93	7.8%
3	Reduce methane from enteric fermentation	0.34	5.00	6.5%
4	Agroforestry	0.26	5.08	4.9%

Table 3. Results of the impact of various farming practices on the carbon footprint and embodied energy input of the farm (changes relative to the base level).

	Base		Relative change compared to the base case (as % of the base case)							
			No-till		Reduced N fertilizer		Anaerobic digester		All together	
	GHG (tCO_2-eq)	Energy (GJ)	GHG	Energy	GHG	Energy	GHG	Energy	GHG	Energy
Tractor fuel	26	491	-0.7	-5.4	0.0	-5.7	-0.6	-4.6	-1.3	-15.7
Enteric fermentation	862	0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Manure management	134	0	0.0	0.0	0.0	0.0	-7.8	0.0	-7.8	0.0
Direct N_2O emissions from soils	167	0	0.0	0.0	-1.4	0.0	-1.4	0.0	-2.9	0.0
Indirect N_2O emissions from soils	14	0	0.0	0.0	0.0	0.0	-0.1	0.0	-0.1	0.0
Electricity purchased (i.e. on the grid)	68	1089	0.0	0.0	0.0	0.0	-5.0	-45.1	-5.0	-45.1
Mineral and organic fertilizers (processing and transportation)	81	559	0.0	0.0	-2.8	-9.1	-3.4	-11.6	-6.2	-20.7
Other crop inputs (seeds, pesticides)	0	275	0.0	-1.5	0.0	-7.9	0.0	-7.9	0.0	-17.3
Total	**1352**	**2414**	**-0.7**	**-6.9**	**-4.3**	**-22.7**	**-18.3**	**-69.2**	**-23.3**	**-98.8**

4. Discussion

In this paper, a fairly simple case study was presented. Perhaps more significantly, the Energy Positive tool also enables comparison of alternative management scenarios on the farm – allowing farmers to see how structural changes to their farm system would influence their energy budget in the future. All of these outputs can be linked to financial data, and given existing concerns about energy costs, this may motivate farmers to take a fresh look at their farm systems. Energy intensity of farms and their products will be of interest to people further along the food chain, in particular retailers, who are increasingly aware of the need to manage risk in their supply chains.

The critical difference between Energy Plus and existing tools is that it focuses on energy and energy cost risk management, not carbon or GHG emissions. Farmers find this much more compatible with their business needs. Energy Plus is a scenario-based strategic planning tool, not a detailed auditing tool. This allows the user to think beyond simple ‘efficiency adjustments’, compare the implications of structural changes to their land management system, and make strategic-level decisions.

5. Conclusion

The results show that synergies between energy risk management and GHG mitigation exist. Therefore, we conclude that focusing on energy risk instead of climate change may be a more effective way of motivating farmers to implement improvements in their farming systems. However, especially in livestock farms with ruminants extra mitigation actions are needed to reduce the GHG emissions from enteric fermentation.

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Consequential and attributional modeling in life cycle assessment of food production systems

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ABSTRACT

The aim is to contribute to a deeper understanding of the differences between the consequential (cLCA) and attributional (aLCA) modeling approaches and to show how the choice between cLCA and aLCA in some cases can be decisive for the outcome of an LCA. We present cradle-to-gate CO_2-eq. emissions (carbon footprint) of barley, soybean meal and palm oil and discuss the differences between the cLCA and aLCA methodologies and results.

Keywords: Consequential LCA, Attributional LCA, Life Cycle Inventory modeling

1. Introduction

Is consequential (cLCA) or attributional (aLCA) modeling the correct choice for life cycle assessment? This is among the most discussed questions within the LCA community during the last 10-15 years. Although an ISO standard was launched several years ago, massive disagreement between LCA practitioners still exists, as the norm does not distinguish between them. The two most important differences between cLCA and aLCA are the handling of multi-functional processes (allocation versus substitution) and whether or not constrained suppliers are included in the market mixes. According to Weidema and Moreno (2013) a comparison between cLCA and aLCA results in ecoinvent showed that more than 12% of the LCIs deviate with more than a factor 2. In this article we discuss the cradle-to-gate CO_2-eq. emissions (carbon footprint) of several food products, namely barley, soybean meal and palm oil. The aim is to contribute to a deeper understanding of the differences between the two modeling approaches and to show how the choice between cLCA and aLCA in some cases can be decisive for the outcome of an LCA.

2. Methods

LCA data on barley, soybean meal and palm oil from Dalgaard et al. (2014) were used to investigate how the modeling approach affects the carbon footprint results. Four different modeling approaches were used to calculate carbon footprint of milk in the study of Dalgaard et al. (2014) and the data are from 2005. The four approaches consist of consequential modeling and three different versions of attributional modeling. However, in the current study, only data from the consequential modeling and the attributional version named ‘allocation/average modeling’ are presented for simplicity. The by-products are accounted for by substitution (cLCA) and price allocation (aLCA). In cLCA only flexible suppliers are included, whereas in aLCA also constrained suppliers are included. Constrained suppliers are suppliers that do not respond to a change in demand for a certain product. In order to reduce complexity, the effects related to indirect land use changes are excluded. For more details on modeling assumptions, see Schmidt and Dalgaard (2012) and Dalgaard and Schmidt (2012).

3. Results

The differences in carbon footprint depending on the applied modeling approach are presented in Figure 1. Both barley, soybean meal and palm oil are products deriving from multi-functional processes. The by-products are the following: Straw from barley production, soybean oil from soybean meal production and palm kernel meal and empty fresh fruit bunches from palm oil production. The allocation in the attributional modeling is based on product prices from 2005. The differences in carbon footprint of barley are and presented in more detail in Figure 2. The cLCA carbon footprint for soybean meal is negative (-0.40 kg CO_2-eq.) because the by-product soybean oil substitutes palm oil on the market (Dalgaard 2008), and the saved emissions from palm oil production more than counterbalance the emissions from soybean production and processing. The difference between the two carbon footprints of palm oil is small. This is mainly because the by-product allocation factors are small in aLCA, due to the low value of the by-products (e.g. palm kernel meal, fresh fruit bunches), so that the oil is attributed most of the impact. In cLCA the by-products have low avoided emissions.

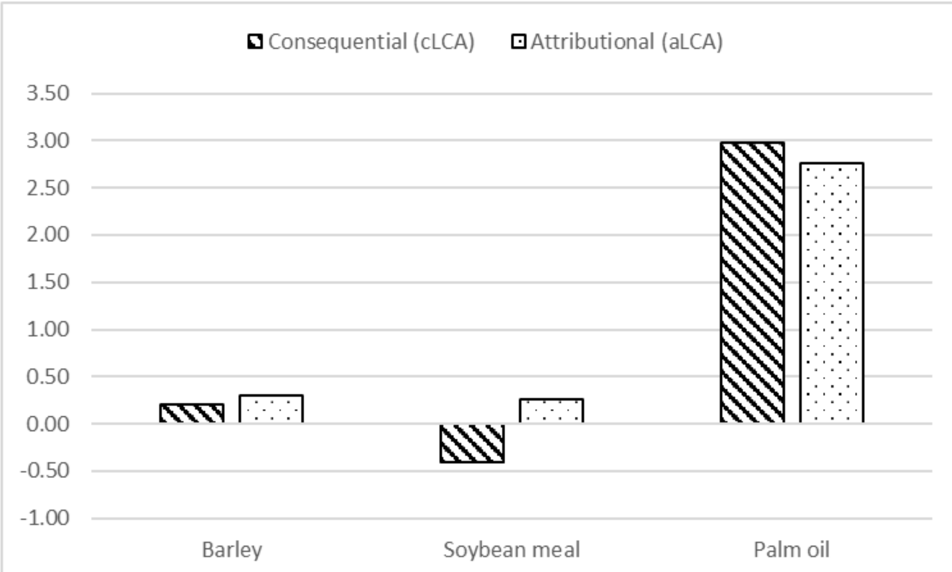

Figure 1. Carbon footprint of barley, soybean meal and palm oil using respectively cLCA and aLCA modeling. Unit: Kg CO_2-eq. per kg.

In this case the production of barley is a multi-functional process from which both barley and straw/energy are produced. According to Statistics Denmark (2012) 75% of the straw was removed from the grain fields in 2005. Barley and straw constitute respectively 75% and 25% of the biological material removed from the field. In cLCA this is handled by substitution. All emissions from the production of barley/straw are ascribed to the main product barley and afterwards the emissions from the saved energy caused by incineration of straw are deducted.

The CFs are 0.21 (cLCA) and 0.30 (aLCA) kg CO_2-eq. per kg barley. The contributions from each part of the product chain are presented in Figure 2. The emissions from the field, fertilizers, diesel and services & capital goods are lower for aLCA mainly because 40% of the GHG-emissions are allocated to straw. However, this is more than counterbalanced by the avoided energy production in cLCA, which is deducted from the barley and reduce the GHG-emission by 0.30 kg CO_2-eq.

However, the emissions related to fertilizer production are 65% lower in the aLCA scenario compared to the cLCA scenario. The main reason for the lower emissions is, as described earlier, that 40% of the emissions from fertilizer production are ascribed to the straw. The remaining 25% of difference between cLCA and aLCA is mainly caused by the differences in fertilizer market mixes used in the cLCA and aLCA. Both manure and mineral fertilizers are applied to the field during the barley cultivation. In cLCA the manure applied to the fields for fertilization is considered as constrained and excluded from the fertilizer mix, whereas in aLCA both manure and mineral fertilizer are included in the mix. The inventory for manure in aLCA is from a multi-functional process from which both manure, beef, milk and energy are produced. Energy is from incineration of fallen cattle. The emissions from the multi-functional process are distributed between the four co-products and only 1% is ascribed to the manure. This results in considerable lower carbon footprint per kg N for manure compared to mineral fertilizer and that contributes significantly to the lower GHG emissions from barley cultivation, when applying the aLCA methodology.

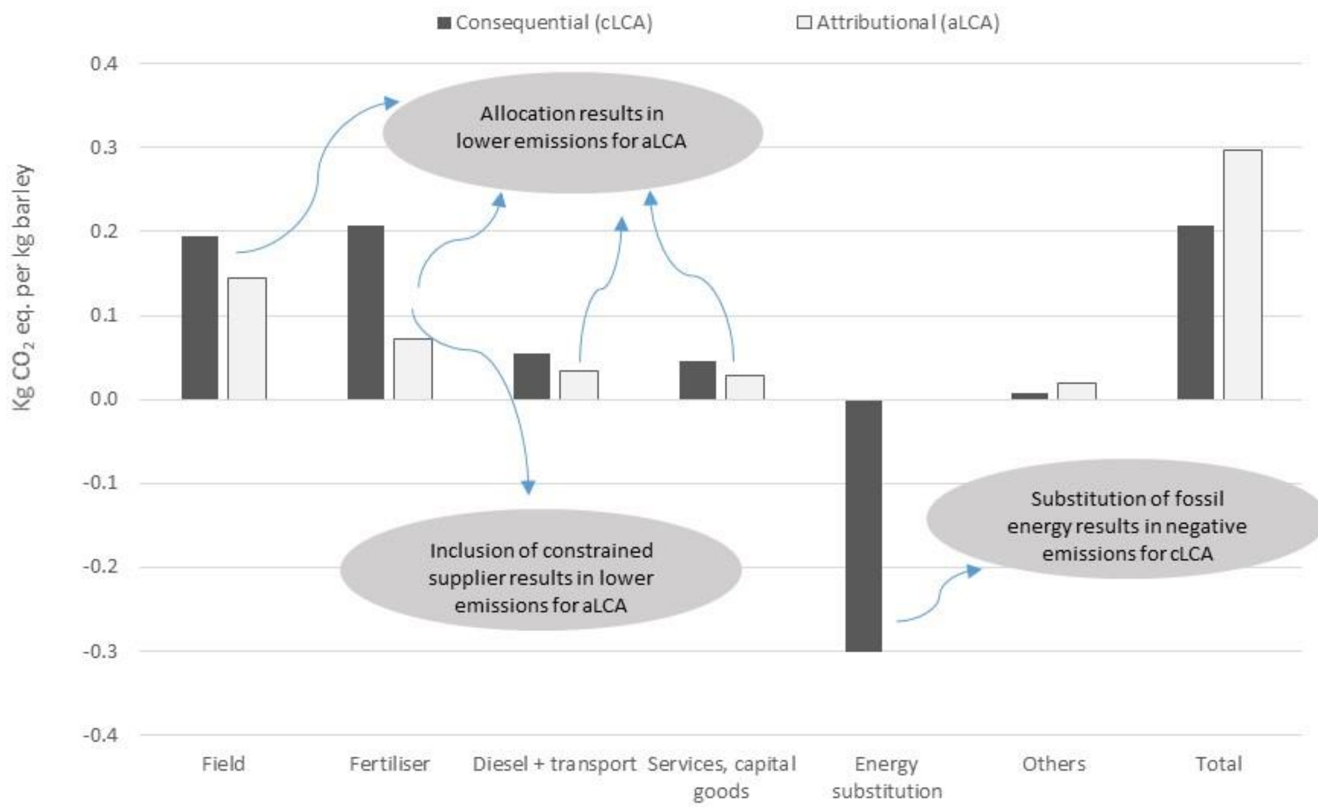

Figure 2. Contribution to GHG emissions from different parts of the barley production chain. Unit: Kg CO_2-eq. per kg barley.

4. Discussion and conclusion

The cLCA and aLCA carbon footprints above show that the main reasons for the differences in results are:

- Substitution (cLCA) versus allocation (aLCA) for handling of multi-functional processes
- Exclusion of constrained suppliers from market mix (cLCA) versus inclusion of constrained suppliers in market mix (aLCA)

Substitution is fundamentally different from allocation. In the barley case it is obvious that the production of barley results in straw production, which again results in saved fossil energy. On the other hand the allocation is a rather artificial way to overcome the multi-functional process issue. In this case, the allocation is based on prices of electricity and heat, but it could as well be straw prices or mass of barley and straw respectively. The definition of allocation method is normative and not based on reality. However, in cLCA the definition of the type of energy substituted can also be challenging for the less experienced cLCA practitioner. If the farmer uses the straw for energy production the carbon footprint of barley will be lower regardless which of the modeling approaches are applied.

Constrained suppliers are excluded in the cLCA modeling, but included in the aLCA modeling. Misleading results might be a consequence of including constrained suppliers in a market mix. In the example with barley the market mix for fertilizer includes manure. If advice to the barley producers was given on the basis of the aLCA, an increased use of manure instead of mineral fertilizer would obviously be a good solution, because the carbon footprint of N in manure is lower than the average N in mineral fertilizer. But in reality, a shift from use of mineral fertilizer to manure will not increase manure production but rather the production of mineral fertilizer. Farmers have cattle because they want to sell milk or beef and not manure, thus an increased demand for manure does not result in more manure production.

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Methodologies accounting for indirect Land Use Change (iLUC): assessment and future development

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ABSTRACT

Land demand is constantly increasing due to increasing population and consumption patterns. When new land is required land use changes are triggered, causing several environmental and social impacts. Particularly debated is the assessment of indirect Land Use Change effects. Several methodological approaches have been proposed for carrying out the assessment. In this paper we reviewed state-of-the-art iLUC models and classified them in two main categories: economic models and deterministic models. Five models have been selected and compared according to fifteen criteria covering modeling framework, impact categories assessed and models' transparency.

The results show that, within a Life Cycle Assessment approach, progresses have been made in Economic General Equilibrium and Deterministic Models and a combination of those can achieve a markedly robust assessment of iLUC. There are still wide margins for improving current models. In particular, there is room for improving precision of data, identification of marginal land and inclusion of a broader range of impact categories.

Keywords: Land Use, indirect Land Use Change, Life Cycle Assessment

1. Introduction

Land demand is increasing due to population growth, demanding food, feed and pasture land. Land use is also caused by demand of fibers, fuel wood and more recently industrial scale biofuels. The result is a constantly rising environmental and social impact. Land use has long been underestimated or ignored in environmental assessment techniques (Lindeijer 2000) but is increasingly being included in impact assessments. Land Use Change (LUC) impacts have recently been introduced in environmental (Banse et al. 2008) and economic analysis (Hertel, Rose, and Tol 2009B). The debate around LUC effects accelerated with the publication of two articles by Fargione et al. (2008) and Searchinger et al. (2008). Until then, the term LUC was mainly referring to direct Land Use Change effect (dLUC), such as greenhouse gas (GHG) emissions, caused by a change in land use, where the change is taking place. Fargione et al. (2008) and Searchinger et al. (2008), instead, investigated the relevance of indirect LUC (iLUC) effects: a change in land use caused indirectly as an upstream consequence of a direct LUC taking place somewhere else in the world. ILUC and dLUC were defined in standards only in 2012 (ISO/TS-14067 2013).

Different approaches and models have been proposed in recent years to solve those controversies but a broad consensus on them still needs to be reached. The controversies include the theoretical framework as well as ways to model the complex global land use dynamics; in particular, difficulties related to: understanding and modeling where land LUC takes place; establishing the relationship between demand of agricultural products and land use changes; accounting for the effect of by-products; the overall level of uncertainties caused by the multiple modeling assumptions (Marelli, Mulligan, and Edwards 2011). The first rather high GHG emissions esteems cause by iLUC rouse a high concern (Fargione et al. 2008, Searchinger et al. 2008); further researches contributed to a progressive downsizing of the estimated effects: in a recent work Dull et al. (2013) show that the estimated LUC GHG emissions from corn ethanol gradually decrease in newer studies. Refined and improved models predict a lower iLUC impact compared to earlier esteems; some authors conclude that iLUC emissions might even be irrelevant (Kim and Dale 2011); other studies still prove how the existence of iLUC effects have neither disappeared nor can be considered as negligible (USA-EPA 2010). Indeed, with the current ethanol and biodiesel production trend, increasing population and pro-capita consumption, it seems difficult to challenge the hypothesis that iLUC is taking place. Tyner et al. estimated that only in the USA a third of corn ethanol production is intended to ethanol (2010). In the meantime, the annual yield growth rates are stationary or decreasing while crop demand is increasing, leading to a constant increase in crop prices (Brandão 2012). Yet, the challenges in estimating the magnitude of LUC and related effects are numerous and models still future a relevant degree of uncertainties, mainly related to data availability and modeling constraints.

The goal of this paper is to underline recent improvement upon models and the aspects that still remain hotspots. Following an extensive review, five models have been chosen to assess their characteristics according to a set of fifteen criteria including: completeness of scope; impact assessment relevance; scientific robustness and certainty; transparency, reproducibility and applicability. For each criterion the models' performances were compared and scored relatively to each other. The data required for reviewing and scoring the models were obtained from scientific literature and models' reports. Since a complete product/process assessment is usually carried out within the framework of Life Cycle Assessment (LCA), this paper reviews iLUC models as part of a more comprehensive LCA framework, in which the modeling results are included, independently of the model used for their assessment.

2. Methods

The review criteria used to assess the performances of the five selected models are reported in Table 1. The criteria were built on the work of Hauschild et al. (2013) modifying the assessment categories and relative criteria to fit the purpose of the current review. The models criteria were grouped in four categories.

Table 1. List of criteria: the criteria are grouped in four categories. Each criterion is meant to answer a specific question reported on the right.

A - Completeness Of Scope	**How in detail does the model covers the environmental mechanisms associated with land use changes?**
i - Dataset	What is the underlying dataset of the study?
ii - Land Classification	Which types of land are classified by the model?
iii - Origin of Marginal Land	How the location of the identified marginal land type is identified?
iv - Co-product	How is the effect of co-product accounted for?
v - Distribution of Emissions	How is the time of GHG emissions associated to the activity?
B - Impact Assessment Relevance	**To what extent are the critical impacts of LUC included and modeled in accordance with the current state of the art?**
vi - Non GHG Emission	Which non-GHG emissions are considered?
vii - Other non-Environmental Impacts	Which non-Environmental Impacts are considered?
C - Scientific Robustness And Certainty	**Does it represent state of the art, can it be validated against monitoring data, and are uncertainties reported?**
viii - Peer-review	Is the model peer-reviewed/presented in peer-reviewed articles?
ix - Uncertainties	Are scenario and model uncertainty taken into account?
x - Updating	How can the model be updated/further developed?
xi - Science-Based	Are the data and assumptions consistent with a science-based approach?
D - Transparency, reproducibility and Applicability	**How accessible are the model, the model documentation and the applied input data? Are the models applicable to different contexts?**
xii - Documentation	Is the model documentation published and easily accessible?
xiii - Input Data	Are the input data publicly available?
xiv - Applicability	Can the model be applied to other contexts/products?
xv - Transparency	How transparent is the reviewed documentation?

According to their main characteristics, existing LUC models have been grouped in three types: Economic Equilibrium Models (EEM), Deterministic Models (DM) and Normative Models (NM); they differ in the extent they address iLUC modeling challenges and the approach used. EEM are divided in Partial Equilibrium models (PE) and Computable General equilibrium models (CGE). Normative Models are simplified approaches, based on rules, established 'norms', not necessary scientifically justified, but based on statistical metadata or normative assumptions. Due to their more simplified nature they have not been considered for comparison with other models. They are however discussed in section 3.

An accurate review following the criteria introduced above was completed for five among EEMs and DMs (Table 2): a modified version of the CGE model GTAP (GTAP 2014, Hertel, Rose, and Tol 2009B) named GTAP-AEZ, the PE model Common Agricultural Policy Regional Impact (CAPRI 2012), the EU Joint Research Center (JRC) hybrid model (Hiederer et al. 2010) and two DM (Bauen et al. 2010, Schmidt et al. 2013). These models were selected after a broad review of the existing literature to cover a wide range of different approaches.

Table 2. Models selected for comparison and description of models' typology

Model	Model type	Selected iLUC Models
Economic Equilibrium model (EEM)	Partial Equilibrium (PE) model	CAPRI
	Computable General Equilibrium (CGE) model	GTAP-AEZ
	Hybrid CGE-PE model	EU-JRC
Deterministic Model (DM)	Participative model	Bauen et al.
	Avoiding type allocation of emissions	Schmidt et al.

GTAP is a global network of researchers coordinated by Purdue University, developing models and databases used also in other CGE models (e.g. IMAGE, LEITAP); CAPRI model is an agricultural sector PE model developed by the European Union (EU) research fund for policy impact assessment in the EU. The EU JRC developed an harmonized spatial dataset integrates data obtained from the CGE model IFPRI-MIRAGE and the PE model AGLINK-COSIMO; the hybrid model expressly assesses GHG changes (including soil) caused by the production of biofuels; Bauen et al. (2010) is a regional DM commissioned by the UK department of transport to assess the GHG emission associated with iLUC caused by biofuel production; through a more transparent and participative approach, it attempts to overcome previous models' limitations by consulting a panel of experts and stakeholders in the development process. The model developed by Schmidt et al. (2013) is a flexible DM, applicable to any land or crop in any location in the world, overcoming a frequently used normative assumption used to distribute emission over time; a similar approach is also proposed by Kløverpris and Mueller (2013).

The performances of the five selected models were scored through a paired comparison analysis: for each criterion, the models were compared against each other and ranked by assigning a comparative value ranging from 1 to 5.

3. Results and discussion

According to the criteria introduced in Table 1, the models were assessed and compared. The results of the comparison are schematically presented in the Table 3.

The final score are expressed as an average and shows considerably homogeneous values, with the exception of the JRC and Bauen et al. models; this is due to the scope of those models, restricted to a regionalized analysis of the biofuel sector. EEMs are generally more robust with regard to the group of criteria 'A': the average score obtained by EEMs for the first four criteria is in fact 2.9 (3.1 excluding the hybrid JRC model), while the average score obtained by the DMs for the first four criteria is 2.5. EEMs show a lower average score when the JRC model is included because the JRC model does not account for the effect of co-products (criterion iv). The calculated average excluded criterion v (Distribution of emissions) from group 'A' because the value for this criterion are rather imbalanced between EEMs and DMs; this is simply due to the fact that EEMs provide an iLUC GHG emission factor but do not suggest a methodology to handle the distribution of the emissions. In the compared EEMs this aspect is left to the results' users. To some extent EEMs consider the issue outside the scope of iLUC modeling and as a general problem to be dealt with anytime emissions' impacts are accounted.

It can be observed that the robustness gained by EEM through the complex methodological framework (group of criteria 'A') tend to be lost in the group of criteria 'D', reflecting a loss of transparency and clarity and traceability of the model. The average score obtained by EEMs in group of criteria 'D' is in fact 2.9 (3.0 excluding the JRC model), while the average score obtained by the DMs is 4. The trend is especially visible in the traceability of the input data (criterion xiii) and overall transparency of the model (criterion xv). The smaller score obtained in criterion xiv (Applicability) by the JRC and Bauen et al. models reflect instead their regional scope and the focus restricted focus to biofuel production. Model uncertainties (criterion ix) are also more difficult to assess in complex EEMs than DMs. On the other hand, because of their complexity, EEMs and the respective databases are generally maintained by large scientific networks or institutions and are more likely to be updated and peer-reviewed.

Group of criterion 'B' shows a very interesting result: at present, the focus of iLUC models (including the compared models) is almost exclusively restricted to GHG emissions impact; among the compared models. CAPRI is the only case in which an attempt is made to assess other environmental impacts than GHG emission, namely, water quality impacts and impacts on biodiversity. There are however other consequences triggered by increasing land use, both environmental (Wicke et al. 2012), social and economic. In life cycle analysis of LUC they are often not included, or only partially (Gawel and Ludwig 2011). The inability to include a full range of impacts caused by iLUCs is a notable limitation of current iLUC models. A more thorough analysis might in fact leads to very different results, suggesting different development patterns than the one indicated by current analysis.

From a LCA perspective, it might be argued that iLUC models concerns data inventory modeling rather than impact assessment and that broadening the range of iLUC impacts is an impact assessment problem rather than an inventory one. Moreover, data collected for quantifying LUC GHG emission (land classes, marginal land, yields etc.) are in some cases the same underling data necessary for a broader impact assessment. If occupation of new land (forest land) and its location are identified for example, it might be relatively easy to assess the consequent loss of biodiversity. Nevertheless, impacts on biodiversity are always pointed out as the most serious when occupation of new land is concerned and comprehensive iLUC models are not consistent if their scope do not included its assessment; yet, to broaden the assessment to further environmental aspects as soil depletion, water use, and non-environmental as social and economic impact, the collection of further data is unavoidable and a the goal and scope of the model needs to be radically reformulated.

Table 3. Results of the paired comparison analysis applied to the five selected models.

A - Completeness Of Scope	**CAPRI**	**GTAP-AEZ**	**JRC**	**Bauen et al.**	**Schmidt et al.**
i - Dataset	3	4	4	2	2
ii - Land Classification	3	5	3	2	5
iii - Origin of Marginal Land	2	2	3	2	4
iv - Co-product	3	3	0	3	0
v - Distribution of Emissions	1	1	1	3	5
B - Impact Assessment Relevance					
vi - Non GHG Emission	2	0	0	0	0
vii - Other non-Environmental Impacts	1	0	0	0	0
C - Scientific Robustness And Certainty					
viii - Peer-review	5	5	3	3	0
ix - Uncertainties	1	3	0	2	4
x - Updating	5	5	3	0	3
xi - Science-Based	4	4	4	4	5
D - Transparency, reproducibility and applicability					
xii - Documentation	5	3	5	4	4
xiii - Input Data	2	1	2	4	5
xiv - Applicability	3	5	2	1	5
xv - Transparency	3	2	2	4	5
Average score	**2.7**	**2.9**	**2.1**	**2.1**	**3.1**

Regarding group of critera ‘C’, EEMs obtain and average score of 3.5 while the average score of DMs is 2.6. Generally, the assumptions behind the models are motivated through science-based argumentations (criterion xi) with the exception of the allocation of GHG emissions flows associated with LUCs over an arbitrary period of time followed by Bauen et al (criterion v). Distributing GHG emission over time is a widely used normative approach but, as such, non-scientifically justifiable. Acknowledging this limitation, Bauen et al. DM calculates the iLUC GHG emission factor considering different allocation time (30 and 100 years). Schmidt et al. DM instead proposes a science based alternative approach, avoiding a normative assumption (criteria v and xi), further discussed in section 3.2. CAPRI and GTAP models have been peer reviewed in scientific journals articles available on the model webpage. The DM by Schmidt et al. is currently under review. Peer-reviewed version of JRC and Bauen et al. models have not been found but their development involved multiple stakeholders and panels of experts. With the exception of Bauen et al. the models are periodically updated but the JRC model does not directly assess uncertainties; yet, the PE and CGE models upon which is based have been both peer-reviewed and show models’ sensitivity to main assumptions.

A further discussion is presented in the following subsections.

3.1. Economic models

EEMs are distinguished between PE and CGE models: PE models represent only a specific subset of economic sectors for a country or region, with no link to other economic sectors, giving an incomplete but rather detailed picture of it. On the contrary, CGE models account for the entire economic flow among different production sector within an economic system (national or global). However, completeness entails an increasing complexity and a loss of details in the model; the representation of land uses, land use alternatives and emissions cannot be sufficiently detailed for modeling land use change. For this reason CGE models are usually expressly modified to suit the model of iLUC. Complexity also adds to CGE models uncertainties difficult to estimate (Dunn et al. 2013).

3.1.1. CAPRI model

Economic models have usually a national level of data aggregation; in some cases efforts have been made to further regionalize models: CAPRI (Common Agricultural Policy Regional Impact) divides the EU territory in about ca. 250 NUTS2 sub-regions (CAPRI 2012). Data are mainly drawn by FAOSTAT, OECD and the Farm Accounting Data Network (FADN). Land use change data are drawn from EUROSTAT land use and land cover data but no particular plausibility check is carried on this datasets. Land uses are classified according to the LUCAS survey, which means 36 crop land classes and 2 of permanent grass rearranged according to the crops present in CAPRI. Since CAPRI has been built to model agricultural activities, forest land, water and urban areas are all aggregated in category 'Other' (criterion ii). Obviously the difference between an urban area and a forest land is rather big. This is of course a limit when modeling LUC. Other than GHG emissions indicator, CAPRI features indicator accounting for ammonia emissions, NPK leaching, water balances, and nitrate concentration in water and chemical emissions. The model also accounts for by-products of land-requiring product by means of physical replacement ratios. Alternatively, other CGE models accounts for by-products by substitution based on relative prices, as the GTAP model, which includes corn ethanol DDGS as a co-product.

3.1.2. GTAP-AEZ model

The Global Trade Analysis Project (GTAP) develops databases and CGE models which have recently seen improvements (GTAP version 8 database has been released in 2012) aiming at assessing LUC impacts. For this purpose it has further been regionalized, linking it to agro-ecological zones (AEZ) model (Fischer et al. 2012): since land types differs, land substitutability is possible only among zones showing similar characteristics (Hertel et al. 2009A). GTAP-AEZ is a special version of the model suited for estimating land use changes. The biofuel sector is here better represented through disaggregation of corn and sugarcane ethanol, and biodiesel, the three major biofuels. Environmental issues, such as climate change, resource use and technological progress in agriculture, are better pictured by dynamic models; GTAP-Dyn, a dynamic version of GTAP (Ianchovichina and Walmsley 2012) aims at projecting future land use change patterns linked at growth rate in each world's regions to identify the marginal land. Forest dynamics are accounted integrating the Global Timber Model (Sohngen and Mendelsohn 2006) while land heterogeneity is limited by the use of AEZ and land mobility across uses by the Constant Elasticity of Substitution (CET) function, estimating land supply elasticity. The CET function simulates the competition for land in economic models through the parameter 'σ', elasticity of land transformation. GTAP database is built upon the integration of several datasets on crop harvested area, production and yields (FAO 2006, Monfreda, Ramankutty, and Foley 2008). However, accounting at global scale for interaction among different economic sectors and regions poses some problems and for this reasons GTAP database aggregates in eight sector all crop productions.

3.1.3. EU JRC model

The hybrid model based on a CGE, PE and geographical information was developed by the Joint Research Center (JRC) of the EU (Hiederer et al. 2010). To calculate the GHG emissions from LUC the model use the cropland demand data acquired by CGE model IFPRI-MIRAGE and the PE model AGLINK-COSIMO. Moreover, the model spatially allocates agricultural land demand: combining different datasets to estimate existing cropped areas, land availability and land suitability they predict the geographically location where the land use change may occur (the location of the marginal land) and create a land suitability map. The model provides precise results with regards to GHG emissions by iLUC since it also includes soil properties information (such as carbon content) drawn from the Harmonized World Soil Database. However, the land demand spatial allocation module also increases the overall complexity, assumptions and consequently, the model uncertainties; at present, uncertainty analyses of the JRC model have not been undertaken (Hiederer et al. 2010). Due to the considerable number of assumptions (economic trends, market variables) and the complexity of model's framework (CET function, prices elasticity etc.), EEM results are rather complex: the extent to which they depend on the assumptions are not intuitively understandable (Nassar et al. 2011). The use of prices and price elasticity to determine yield change and cropped area variations in EEM does not reflect the role that other factors, such as technological and infrastructure constraints, yearly harvest capacity, trade agreement and other factors may play in determining those changes; even when these factors affect prices, price response is not visible in the short term. Yet, the complexity of the EEMs does not broaden the spectrum of the analysis and aspects as land degradation and biodiversity are generally not included in these models.

3.2. Deterministic Models

Alternative approaches to EEMs are causal-descriptive or Deterministic Models. DMs provide a simpler and more transparent framework than EEM where those data can be applied. They describe the future states of a system based on cause-effect relationships observed through statistical and historical data. They tend to have a more simple approach than EEMs (Nassar et al. 2011). That reduces the computational effort, data requirement and makes them conceptually easier to understand. Nevertheless, DMs do not necessary exclude economic aspects driving the supply/demand patterns: in some cases DM also includes market information such as cross-price elasticity of products, making them hybrid models, able to draw from different approaches. According to current market trends and assumptions on agriculture supply/demand trajectories, they forecast future production and consumption patterns. From this scenario, future land uses and origin are estimated.

3.2.1. Bauen et al. model

On behalf of the UK's Department of Transport Bauen et al. (2010) developed a DM to assess iLUC from five biofuels, taking into account future yield and area changes' trends, products substitutability and the effect of co-products; data were extrapolated from databases through statistical analysis of historical trends. A peculiar characteristic of this study is that future predictions were estimated consulting a panel of expert and stakeholders, to integrate their views in the assessment and guarantee a more democratic approach compared to the economic models' approach. Cederberg et al. (2011) used a deterministic approach to assess the impact of beef production in Brazil. This regional focus and product-oriented model make it unfit for application to different contexts, a limit generally observed in many DMs. As Bauen et al. (2010), also Cederberg et al. only account for iLUC GHG emissions effects. Within the LCA framework DMs are also distinguished between models using averages of historical data and future oriented models; the latters may also use historical data but to predict future trends, which can also diverge from historical patterns. This difference is also reflected in the LCA approach chosen, Attributional (A)LCA or Consequential (C)LCA (Sanchez et al. 2012). The dispute between ALCA-CLCA is beyond the scope of this paper; however, thorough discussions upon details of the two approaches are present in the scientific literature (Weidema 2003, Schmidt 2008, Thomassen et al. 2008, Finnveden et al. 2009, Rehl, Lansche, and Müller 2012).

3.2.2. Schmidt et al. model

A limit of iLUC models, regardless of the approach used, is the time allocation of the emission over an arbitrary period of time, generally 20 years (Fritsche, Hennenberg, and Hünecke 2010, IFPRI 2010) or 30 years (Bauen et al. 2010). Since this choice has no scientific bases, Schmidt et al. proposed an alternative DM approach, avoiding time allocation of emission, similar to the approach proposed also by Kløverpris and Mueller (2013). Kløverpris and Mueller model (2013) is applicable to any phenomenon causing LUC, based on current trends and future predictions of land uses. The starting point is the principle that land uses baseline is dynamic rather than static therefore in a region with expanding agricultural area, further occupation of land is seen as "accelerated expansion" and in a region with decreasing agricultural area as a "delayed reversion" (Kloverpris and Mueller 2013). As Kloverpris and Mueller (2013), Schmidt et al. (2013) avoid time allocation of emission considering an increasing land demand as a dynamic baseline; the result is only a time shift in the emission profile and consequently in the measured Global Warming Potential (GWP) effect. Doing so, the GWP is caused from the accelerated expansion of land and the resulting effect is accounted only in the specific year the activity 'land occupation' is taking place; therefore, the emissions do not need to be allocated over an arbitrary period of time. In accordance with a LCA framework Schmidt et al. see "Land" as an input to a process creating an output (the product), as in LCA is treated any other process's input. Schmidt et al. (2013) model is a more comprehensive model than Kløverpris and Mueller (2013). Based on global statistics (FAOSTAT, FAO) and IPPC guidelines, the model is applicable to all crops and regions in the world. Despite being a DM, it does not refrain from drawing upon different toolboxes: economic tools, such as cross-price elasticity tables, are used to identify product substitute and calculate the related environmental consequences (e.g. GHG emissions impact measured by GWP). The authors draw from bio-physical relations and statistical data to identify the marginal land and the share of land intensification. The model assumes that changes in land uses are a consequence of change in land demand; land is treated as any other LCA input and land productivity (the reference flow) is measures in Net Primary Productivity (NPP) (Haberl et al. 2007). The result is DM tending towards a hybrid approach. The limit of this approach is that its validity is limited by the hypothesis that land demand is constantly increasing worldwide: despite at present this is globally a valid hypothesis, it will inevitability fall at some point due to the fact that land is a constrained resource and, if not contained earlier, land demand will be in the end constrained by land availability.

3.3. Normative Models

The last type of modeling approach is the most simplified Normative Model explained above. Flynn et al. (2012) for example proposed a NM based on IPCC national GHG inventory methodologies to assess LUC impacts from crops. The model is intended to be applied in context where complex process-based spatial models to feed agro-economic models are not available, such as in developing countries, and when information on crop origin and growing conditions is limited. The model allows to convert per ha emissions from LUC to a per-ton of product basis (Flynn et al. 2012) but the analysis is restricted to the top 20 producing countries of the assessed product, and a single yield value is assumed for each county. Generally, NMs are intrinsically based on normative assumptions, regardless of whether those being scientifically justifiable or not. NMs are hence suitable for simple applications, such as for illustrative or didactical purposes but are not suitable to support policy making or decision making processes, where more accurate, scientific-sound analysis are required.
A normative approach is also adopted in calculating the iLUC factor by Fritsche et al.(2010), obtained dividing the share of land for biofuel production in 25% as coming from intensification of "set free" land, with a zero displacement risk, and 75% from "new land" representing the actual iLUC factor. This simplified solution to quantify the GHG emissions associated to iLUC is paid by a least reliable result since the factor is calculated ignoring the complexity of cause-effect relationships triggered by LUC. Normative assumptions can be found also in non-normative models: the distribution of GHG emissions over an arbitrary period of time of 20 or 30 years described above can be found for example both in analysis using DM and EEM. Searchinger et al. (2008) for example, used for their model the FAPRI-CARD PE model but assumed bioenergy as the only source of iLUC, since ethanol is a relatively new product demanding a large amount of land, therefore responsible for increasing the global land demand compared to usual trends. This normative assumption is not completely true as land demand is caused by any conventional agricultural activity (Nassar et al. 2011) as well as increasing demand of conventional agriculture products, urban sprawl, forest fires, logging etc.

4. Conclusion

There are very diverse solutions to model iLUC in literature. Considerable progress has been made to adapt EEMs and DMs to capture the complex dynamic of land use changes. Transparency is usually inversely proportional to complexity but complexity of models is to some extent necessary to compensate for the lack of primary data. New studies aiming at collecting global land use and yield data will in the future help reducing complexity. Currently few hybrid economic models, linked to bio-physical and geographical models, can provide scientific sound and accurate picture of iLUC. They are however complex, and computationally intensive and are therefore recommended for large scale applications, such as policy making. For small scale applications, such as LCA of product or processes for companies or small institutions, DMs provide a more useful picture of the cause-effect relationships and product chain. Time allocation of emissions can however be avoided also by EEM using the approach proposed bt Schmidt et al. (2013). A substantial limitation of current model is the inability to include the full range of impacts caused by iLUC since the focus is generally restricted to GHG emissions impact. More comprehensive analysis might show very different results and suggest completely different development patterns. It is therefore in this direction that future research should be addressed.

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Key Environmental Performance Indicators for a simplified LCA in food supply chains

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ABSTRACT

The European research project SENSE will deliver an affordable and comprehensive environmental evaluation system oriented to small and medium enterprises (SMEs) in the food and drink sector. The system aims at simplifying data collection and information requirements by compiling key environmental performance indicators (KEPIs) throughout the value chain in a web-based tool. Cradle-to-gate life cycle assessments (LCA) were conducted in three selected food sectors, namely meat and dairy, fruit juice and aquaculture salmon, in order to verify, if the KEPIs that had been selected for the tool contributed to the main environmental impacts. Since the impact categories cover a wide range of environmental impacts such as acidification, climate change, eutrophication and toxicity, the KEPIs can be used to deliver a comprehensive environmental evaluation. An overview of the main results of the LCA studies and justification for the selection of KEPIs is presented. Moreover, aspects of regionalization of background database is discussed since this may affect the results and needed to be considered when designing the simplified SENSE tool.

Keywords: LCA, key environmental performance indicators, food and drink supply chains, regionalization

1. Introduction

Life cycle thinking and taking responsibility in environmental issues beyond the operation of the companies is gradually being implemented in large businesses along with the awareness of the concept sustainability. This trend is less pronounced in small- and medium sized enterprises (SMEs), often because of lack of understanding and limited capacities to look beyond their daily operation. Moreover, SMEs may not be familiar with environmental assessment methods. Measures to assess environmental performance need to have a positive economic impact, since data gathering is often regarded a burden and companies therefore are not willing to undertake such an assessment. However, when given opportunities to implement life cycle tools, there appears to be potential incentives in SMEs to use Life Cycle Assessment (LCA) results to create an image for the product and the organization, to use in marketing, and for product development (Witczak, 2014). The SENSE project aims at enhancing environmental awareness in SMEs in the food sector by offering a harmonized data collection system and simplified assessment of environmental impacts.

The main environmental challenges of European food and drink supply chains and their environmental impacts have been assessed in the project based on earlier LCA studies (Aronsson et al., 2014). The approach has been to select a set of harmonized input data, defined as key environmental performance indicators (KEPIs) which are essentially the required information for LCA (i.e.: water, energy, materials consumption). Full scale LCAs for three selected food and drink supply chains were conducted in the project and the result interpreted to justify the validity of the selected KEPIs to reflect the main environmental challenges. Key attributes and suitable scope of essential input data was thus prioritized according to the most important environmental impacts in order to simplify data collection in SMEs. The KEPIs are either common performance indicators such as electricity consumption, water consumption, fertilizers and pesticide use, but also key parameters such as the composition of the feed. The results from the LCA studies presented herein thus confirm the most relevant stages in the life cycle of the respective food products and the suitability of the KEPIs to be applied in the SENSE tool.

2. Methods

Three LCA case studies were performed on current food production and supply systems and investigated from a regional perspective:

- orange juice production in Spain (Doublet et al. 2013a)
- dairy and beef production in Romania (Doublet et al. 2013b)
- salmon aquaculture in Iceland and smokehouse in France (Ingólfsdóttir et al. 2013)

Further information on the definition of goal and scope, the life cycle stages included, definition of the system boundary, input materials/items included and excluded, justifications and assumptions made, detailed life cycle impact results and interpretation are available in the respective reports. The environmental impact assessment methods initially selected by the SENSE project team comply with the ones later recommended by ILCD (JRC, 2011). These are the same methods as later recommended by the European Commission on the Product Environmental Footprint (EC, 2013) and in the ENVIFOOD protocol except for water depletion where a revised approach to water footprinting is recommended in the ENVIFOOD protocol (ENVIFOOD, 2012).

In the LCA case studies the allocation for the aquaculture and the orange juice supply chains followed economic allocation as recommended by the ENVIFOOD protocol (ENVIFOOD, 2012). The allocation for meat and milk at the dairy farm followed the physical allocation approach suggested by the International Dairy Federation (IDF 2010). The allocation for the meat produced at the slaughterhouse followed an economical approach, while the allocation for the dairy products at the dairy plant follows a physico-chemical approach as suggested by IDF (2010).

3. Results

The results of the life cycle impact assessment in the three food supply chains analyzed by LCA show similarities as expected. The cultivation of biomass was the main contributor in the orange juice and beef and dairy food supply chains due to the environmental impacts of use of water, land, pesticides and fertilizers, and fuel for tractors. This is also the case for the aquaculture food supply chain including the feed from marine resources where the fuel use for vessels in fisheries is a significant contributor (Figures 1-3).

3.1. Life Cycle Assessment - Orange juice

The impact assessment of the Not-From-Concentrate orange juice shows that the main contribution of life cycle step depends on the impact categories assessed. About 50 % of the climate change and abiotic resource depletion are due to the bottling process.

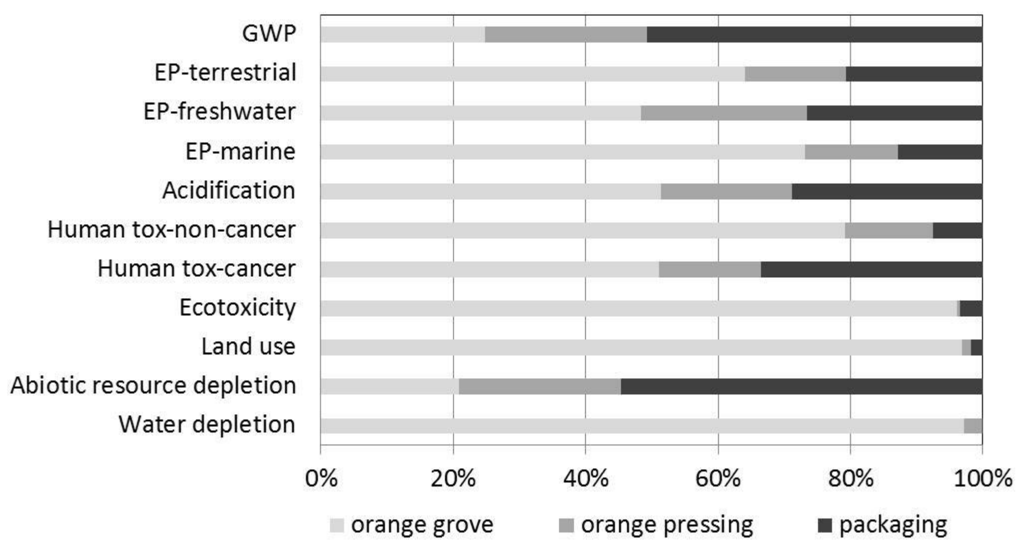


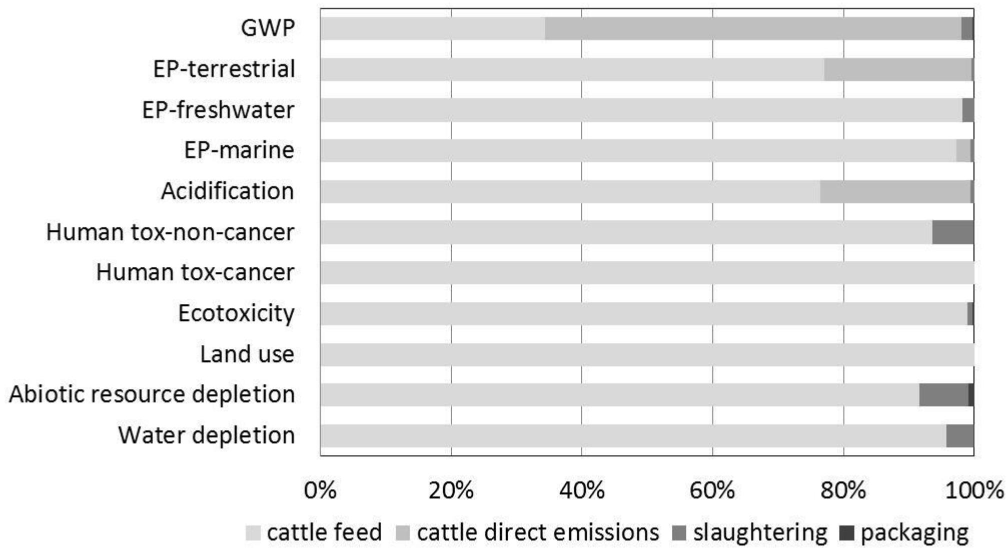
Figure 1. LCA of orange juice in Spain. Impact assessment of one litre NFC orange juice (Doublet et al., 2013a)

The impact categories land use, water depletion and freshwater ecotoxicity are dominated by the orange cultivation (more than 95 %). The orange cultivation contributes around 50 % to the acidification and freshwater eutrophication. The four main contributors to the orange cultivation are the electricity use for the irrigation, the production and use of fertilizers and the application of pesticides.

The most relevant processes for the juice pressing are the electricity use and thermal energy use. The main contributor to the bottling process is the manufacture of the PET bottle.

3.2. Life Cycle Assessment - Beef and dairy

The impact assessment of the beef shows that the feed cultivation at the dairy farm is the main contributor to the results (Figure 2). The slaughtering process and the packaging are negligible to most of the impact categories.



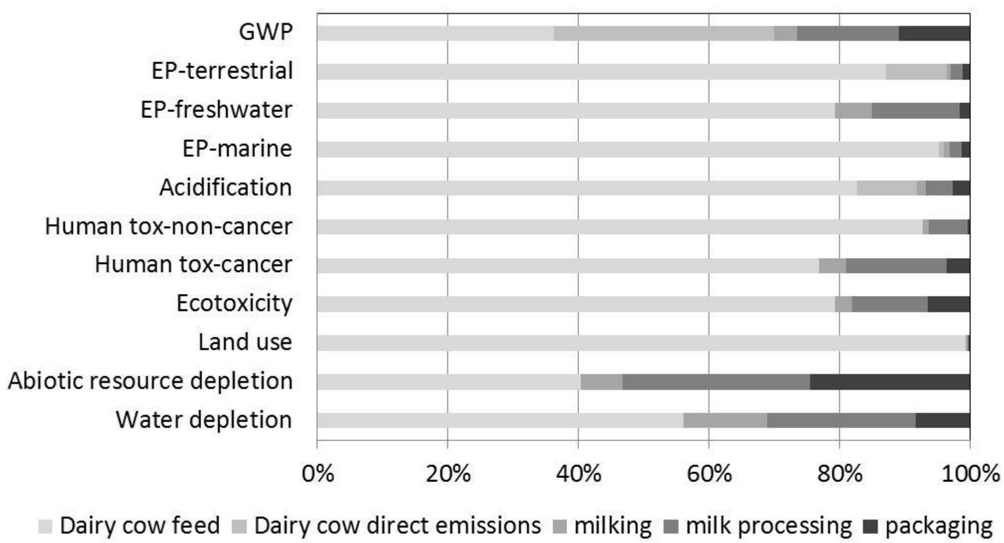


Figure 2. LCA of beef and dairy products in Romania. Impact assessment of 1 kg beef at slaughterhouse (above) and impact assessment of 1 liter milk at dairy plant (below) (Doublet et al., 2013b)

The emissions from the use of fertilizers, manure and diesel for the agricultural machinery influence the results most. The cattle emissions due to the enteric fermentation are the main source for the climate change. The animal waste disposal from slaughtering is also an important step due to its processing into animal flour before its incineration. The impact assessment of the dairy products is similar to the beef because raw milk is produced by the dairy cows. The dairy farm is also the most important step to most of the dairy products. However, the contribution of the processing step to the production of dairy products is higher than the contribution of the slaughtering process to the beef production.

3.3. Life Cycle Assessment of aquaculture

The impact assessment of fresh salmon (head on gutted) transported from Iceland to Europe by sea freight verified that the feed production is by far the dominant life cycle stage in all impact categories. For most of the impact categories (GWP, terrestrial eutrophication, freshwater eutrophication, acidification, human toxicity potential (cancer effects), ecotoxicity, resource deletion and water depletion) this is due to the harvesting and processing of feed ingredients (marine and crop).

For the marine eutrophication impact category it is the release of organic matter to sea (feces, uneaten feed and dead fish) which is the major cause of impact at the farm and for the human toxicity potential (non-cancer effects) the main contribution is the transportation of the feed from feed mill to the farm and long distance distribution of products.

The impact assessment of smoked salmon fillets where fresh salmon (head on gutted) is transported from Iceland to Europe by sea freight and further processed in France showed that for nine impact categories the aquaculture farm life cycle stage is the main contributor of environmental impacts, mainly due to the feed. In two impact categories the human toxicity potential (non-cancer effects) and the water depletion the operation of the smokehouse in France is the main source of impact (Figure 3).

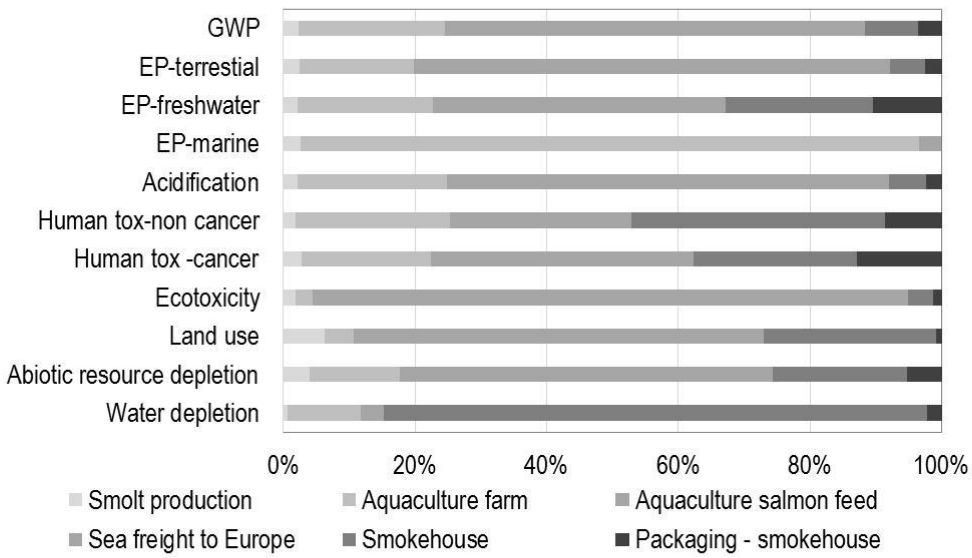


Figure 3. LCA of aquaculture Iceland and smokehouse in France. Impact assessment of 1 kg smoked salmon fillets (Ingólfsdóttir et al., 2013)

3.4. Key Environmental Performance Indicators (KEPIs)

The key environmental performance indicators were proposed as simple-to-measure indicators that could be used as input data in the SENSE tool to calculate the environmental impacts. The criteria for selection of input data for the SENSE tool, was the contribution to the main environmental impacts in the respective life cycle stage. The LCA results thus confirmed the validity of the selected KEPIs to be applied in the SENSE tool based on their relevance for the environmental impact, the data availability and the ease of measurement.

The KEPIs have been grouped according to the different production steps in the supply chain and are presented in a table for each production step. The following production steps were defined:

- Plant production (food and feed)
- Fisheries
- Aquaculture
- Livestock

- Food and feed processing
- Transport

Impact category	Plant production								Fisheries	Aquaculture						Livestock - ruminants					Food and feed processing					Special			Main pollutants / Challenges
																										Dairy	Slaughtering	Juice processing	
Unit	kg N/hectare kg N/kg crop	kg-P2O5/hectare kg-P2O5/kg crop	kg N/hectare kg N/kg crop	kg/hectare kg/kg crop	l/hectare l/kg crop	ha/kg crop	ha/kg crop	m3/hectare m3/kg crop	MJ/kg product	kg feed /kg fish	MJ/kg product	kWh/l product	kg waste/kg product	m3/kg product	kg/kg product	kg raw milk/dairy cow	kg feed/kg live weight	m2/kg product	kWh/kg raw milk	m3/ kg raw milk	MJ/kg product	kWh/l product	m3/ kg product	type/kg product	kg waste/kg product	kg raw milk/kg product	kg live weight/kg meat	kg orange/l orange juice	
Key Environmental Performance Indicator (KEPI)	N-fertiliser use	P2O5-fertiliser use	Manure and slurry application	Pesticide and active substance content	Diesel use incl. machineries	Arable land use	Grazing land use	Water use	Energy use	Feed Efficiency (FCR)[1]	Energy use	Electricity use	Organic waste to sea	Water use	Packaging material	Raw milk production	Feed efficiency	Buildings	Electricity use milking	Water use milking	Energy use	Electricity use	Water use	Packaging material	Waste	Raw milk input	Meat production	Yield	
All impact categories										x						x	x								x	x	x	x	
Climate change	N2O		N2O		CO2				CO2		CO2	CO2			CO2	CH4			CO2		CO2	CO2		CO2					CO2, CH4, N2O
Human toxicity		HM			HM													HM											Heavy metals (HM)
Acidification	NH3		NH3		NOx											NH3													NOx, NH3
Eutrophication,	NOx		NH3		NOx																								NO3, NH3
Eutrophication,		PO4		PPP	PO4	PO4	PO4																						PO4
Eutrophication, marine	NO3		NO3		NOx								x																NO3 (Nitrate), NOx
Ecotoxicity, freshwater		HM		PPP	HM																								Heavy metals (HM)
Land use						x	x																						Land use (m^2 and type)
Abiotic resource	x	x		x	x				x		x	x			x			x	x		x	x		x					Fossil resources
Water depletion					x			x						x						x			x						Water use

[1] FCR = Feed Conversion Rate: Feed used/Fish produced

Figure 4. Selection of Key Environmental Performance Indicators (KEPIs) for each production step for the three investigated supply chains in the SENSE project (Landquist et al., 2013)

The KEPIs selected for the production of all the food supply chains are shown in Figure 4. Each KEPI is given a name and a unit. When the contribution of the KEPI to an impact category is relevant, the cell is shaded and either filled with a cross or with the main pollutant emitted by the KEPI, e.g. carbon dioxide, heavy metals, ammonia, phosphate, etc. The selected KEPIs covered 95%, on average, of the environmental impacts of the respective food supply chains.

Allocation factors were computed on the basis of the shares of the different output products in the turnover, which were given by the plant. Hence, the "shares of products in turnover" is a KEPI.

It should be noted that most of the KEPIs are relevant for many more food products since most food supply chains have common characteristics and similar processing steps. For example the KEPIs for Plant production and for Food and feed processing are the same for many food chains. In all cases KEPIs were adjusted to fit general supply chains although these were not identified as a KEPI in the case studies. There are also some indicators that are specific for some production step, e.g. feed composition and feed conversion ratio (FCR) are specific to the aquaculture chain and the livestock.

3.4.1. Plant production

The plant production corresponds to the crop cultivation including the crop for feed (livestock and aquaculture) and orange cultivation. Plant production requires the use of fertilizer, manure, liquid manure, pesticide, agricultural machinery, land, water and storehouse. It is vital to know the composition of the feed as the impacts differ for the different ingredients.

The KEPIs N-fertilizer use and P_2O_5-fertiliser use refer to the production and use of these fertilizers. It is important to differentiate between the types of fertilizers applied on crops. The N-fertilizer has a higher contribution to the climate change, acidification, terrestrial and marine eutrophication whereas the P_2O_5-fertiliser contributes mainly to the human toxicity, freshwater ecotoxicity and the freshwater eutrophication. Therefore, it is important that the farm informs separately the amounts applied of each type of fertilizers. The production and the emissions to air, soil and water are covered in the background system and not asked to the farm.

The KEPI manure and slurry application refers to the manure and liquid manure (or slurry) applied on crops and are determinant for the climate change, acidification, terrestrial and marine eutrophication. The farm provides the application rate and the emissions to water and to air are included in the model.

The KEPI pesticide and active substance content includes the production of pesticides and the emissions from the active substances contained in the pesticides applied. It is important that the farm provides the pesticide name and the content of active substance. If the latter is not known, it can be found in literature from the pesticide name. The active substances are necessary to estimate the emissions that affect the freshwater ecotoxicity. In a similar way as for the fertilizers, the production of pesticides is included in the background system.

The KEPI diesel use incl. machineries, refers to the diesel consumption including its production and the agricultural machineries used. The diesel production and the emissions resulting from use are included in the background system. The CO_2 emissions due to combustion of fuels can be directly calculated with the amount of fuel burned. The agricultural machinery fleet contributes to the human toxicity and eco-toxicity impacts as well as the freshwater eutrophication. It can be difficult for a farm to estimate its agricultural machinery fleet. It was suggested to have an estimation of the agricultural machinery as a background process linked to the diesel consumption, as it was done also for the case study. Indeed, the diesel use for the agricultural processes is modelled with a dataset that includes the diesel fuel consumption, the corresponding amount of agricultural machinery needed (tractor, trailer, harvester, tillage) and its production and the shed corresponding to the machinery use.

The arable land use and the grazing land use are KEPIs for the land use impact category. The emissions of phosphorus to water due to land use affect the freshwater eutrophication impact category.

The direct water use is also a KEPI related to the water depletion impact category.

The construction of the farm buildings affects mainly the human toxicity due to the impacts of the construction material production. The farm should provide the area of the storehouse, but office buildings can be omitted based on the experience in this case study. It has to be noted that in the current case study on meat production the fodder was produced on the same farm as the animals. If animal feed is bought on the market the relevant KEPIs have to be investigated for the production of all the different type of feed bought by the farm, e.g. soy bean, maize, by-products of food and bioenergy production etc.

3.4.2. Fisheries

The feed used at the hatchery and aquaculture farm consists of marine and crop ingredients. The energy use at the fisheries was identified as KEPI for the aquaculture salmon supply chain since fossil fuels have a high contribution to climate change, acidification, human toxicity, terrestrial eutrophication and abiotic resource depletion.

3.4.3. Livestock

The farm should provide the herd size and the shares of dairy cow, bull, calf, suckler cow. This is covered in the KEPI herd size. It determines the fodder production, the manure and slurry production and the milk and animals sold to the slaughterhouse. By giving the cattle average weight in each category and the raw milk production together with its fat content and protein content, the livestock emissions of methane and ammonia can be estimated. This is the reason why these three parameters are considered as KEPIs. The feed efficiency is also an important parameter to compare different kind of feeding system. This information was however not available for the beef case study.

The KEPI animals sent to slaughtering and raw milk production together with the protein and fat content are needed for the allocation approach. The farm should also give the area of the cattle housing. The construction materials affect mainly the human toxicity and ecotoxicity.

Milking: The KEPIs electricity use and water use were identified for this process. The electricity use covers the electricity production. However, environmental impacts of electricity production vary from country to country. Consequently different impact categories might be affected by the electricity use if case studies are elaborated in another country.

Slaughtering: The KEPIs meat production and meat waste should be provided by the slaughterhouse in order to allocate the slaughtering process to the production of beef and assess the meat yield, i.e. the ratio of meat per livestock animal. The case study on meat includes the slaughtering waste treatment with background data.

3.4.4. Aquaculture

The aquaculture includes both the smolt production and the salmon farming. The KEPIs identified for the smolt production are feed, water use and electricity. The composition and amount of feed used are important in terms of climate change, human toxicity, acidification, eutrophication (terrestrial, freshwater) and ecotoxicity. Information on the total production at hatchery as well as at the aquaculture farm and share of products in turnover are also identified as KEPIs for allocation purposes.

For the aquaculture farm the feed composition and amount, organic waste to sea, electricity use, fossil fuels, water use and packaging materials are identified as KEPIs. The slaughtering and primary processing (gutting) are included in aquaculture. Water use and electricity use are identified as KEPIs even though these indicators are not significant in the net pen aquaculture system analyzed in this study However, these KEPIs can be very important for other aquaculture systems for example land based systems in other regions where renewable energy sources are not available and water is scarce. The other KEPIs affect climate change, human toxicity, acidification, eutrophication as well as abiotic resource depletion. The feed efficiency (FCR, feed conversion ratio), i.e. the weight of feed used (kg) compared to weight of fish produced (kg) is a key factor to assess environmental performance of the aquaculture farm. Furthermore, to assess the performance of aquaculture farms, the amount of marine resources that is consumed in the production of farmed fish, the FIFO ratio (fish in - fish out) is commonly used in the industry.

Feed intake and feed efficiency may change during the lifetime of the farmed fish. In this study the feed conversion ratio was available from the company based on the annual production. However, because the life cycle of salmon is 2-3 years, the annual data may give misleading information about the actual feed conversion ratio. In order to avoid this variation affecting the LCA results it may be more appropriate to use a three year average for the operation of aquaculture farms.

It is important to mention that the aquaculture farm in this study does not use anti fouling agents. Therefore, it is possible that in the case where anti fouling agents are used that they can be of importance. Furthermore, land based aquaculture was not analyzed in this study.

3.4.5. Feed and food processing

Dairy plant: The dairy plant does not only comprise the infrastructure but the whole process of transforming raw milk into dairy products. The raw milk input as well as the electricity use, thermal energy use and water use can be given on a whole of factory basis and allocate to each dairy product thank to the allocation approach applied, e.g. the IDF matrix (Doublet et al., 2013b). Furthermore the produced amount of each single dairy product has to be reported. The infrastructure of the dairy plant could be included in the background data related to the raw milk input processed at the dairy plant. It is also important to know the packaging material and its weight especially for the milk PE bottle and the yoghurt. The production of the packaging should be included in the background system.

Smokehouse: For the smokehouse the following KEPIs are identified: electricity, fossil fuel and water use as well as raw material inputs (salmon HOG), total production and share of products turnovers. The electricity and fuel have impact on climate change, human toxicity, eutrophication (terrestrial) and abiotic resource depletion. Water use has potential influence on water depletion; depending on the region the process takes place in. For land use impacts, the use of wood chips for the smoking process can also be of importance as a KEPI. The head and bones from filleting process as well as cut offs and trimmings from finished products are discarded and do therefore not carry any environmental burden in this study. This may however be of interest if sold as added value by-products.

Orange Juice processing: The juice processing plant must provide the input mass (kg) of oranges needed to produce 1 l of orange juice. The electricity use, the thermal energy use and the water use are the three main KEPIs. Both electricity and thermal energy contribute to the abiotic resource depletion and the climate change.

The water use determines the amount of wastewater that will be treated. The phosphate emissions resulting from the wastewater treatment affect the freshwater eutrophication.

There are by-products from the orange juice processing, e.g. peels, pulp, and essential oils. An allocation approach is necessary to allocate the energy and material flows to the orange juice. In our case study, the allocation factors were computed on the basis of the shares of the different output products in the turnover, which were given by the plant. Hence, the "shares of products in turnover" is a KEPI.

Bottling process: In most cases, the bottling plant does not only bottle orange juice. Therefore, the share of orange juice in the total amount of juice processed is a KEPI necessary to allocate the energy and material flows to the orange juice. The KEPIs electricity use and the thermal energy use cover the energy consumption of the bottles dryers and blowers, compressors, labelling machines, palletizers etc.

The environmental impacts of the KEPI "type of container" depend on the packaging investigated. In our case study, the packaging investigated is a PET bottle. The KEPI includes the PET material production, the PET granulates injection molding into PET preforms and the production of other materials that are included in the PET bottle e.g. secondary packaging, intermediate layer etc. It is relevant for the abiotic resource depletion, the human toxicity, the climate change, the acidification and the freshwater eutrophication. All these processes are included in the background system but the weight of the PET bottle and the other materials must be provided by the bottling plant.

3.4.6. Transportation

In the case study on meat the transportation distances between farm and further processing were quite small. There are also not major transports of fodder products to the farm. Therefore impacts due to transportation were not found to be a major issue and are not considered in the definition of KEPIs for the meat and dairy chain. This conclusion is however not valid for cases with higher transportation distances involved between processing stages as is the case in the aquaculture chain. For transportation the KEPI identified is transportation mode and distance travelled. The fuel is important factor in terms of climate change, acidification, human toxicity, eutrophication (terrestrial) and abiotic depletion.

4. Discussion

4.1. Regional characteristics and background database system

An important question of the project is the adjustment of the SENSE model to regional characteristics. The regional variation affects some of the identified KEPIs and the environmental impacts.

In the SENSE web-based tool the background information is based on the ecoinvent database and is not under the direct influence of the SME. Regionalization of background data is important when designing the simplified SENSE tool, since this may affect the results.

In many cases LCI background data are just available for a global or a European production mix. But, in practice the markets in different regions might be supplied with a different mix of products. Thus, also LCI data can be adapted to the market situation in a specific region. One example of regionalization of background LCI data is the application of a country-specific electricity mix. Publicly available country-specific electricity mix datasets have been implemented in the SENSE tool as background data.

Other important regionalized impact assessment methods were included in the SENSE tool such as water depletion, acidification and terrestrial eutrophication. For acidification regional characterization factors for many countries in Europe are available (Posch et al., 2008). Acidification characterization factors for sulfur dioxide, nitrogen oxides and ammonia are available for France and differ somewhat from the weighted average factors used in this case study.

In the context of the current LCA study on smoked salmon in France, terrestrial eutrophication, regionalized characterization factors are available for France which are higher than the weighted average applied in this study. Furthermore, for marine eutrophication it is important that datasets for emissions of organic matter to sea are available in the SENSE tool. Background LCI datasets need to be available on the nitrogen content in different aquaculture fish species to be able to assess the marine eutrophication potential from dead fish. Additionally, datasets should be available in the tool for N content from feces and feed deposition for sea based aquaculture in different regions. For other background data it was not expected that including regionalized data for diesel, natural gas, fuel oil would make major differences in the environmental impacts results.

Availability of water differs greatly between countries and regions. Regional characterization factors are available for water scarcity (Frischknecht et al., 2009). In this case study regionalization factors for water depletion were used. The smokehouse in France has considerably higher impact on water depletion than the hatchery and aquaculture farm in Iceland, although they use significantly higher amount of water. This is because water is defined as abundant in Iceland.

The feed for aquaculture is composed of both crop products (e.g. soya, rape, wheat) and marine ingredients. The use of diesel for fisheries is the main contributor to CO_2 emissions contributed by the use of fish ingredients in the feed and this varies depending on the type of fishery. Often the information on the feed ingredient composition is not publicly available. It was recommended that Life Cycle Inventory (LCI) background data for different types of aquaculture feed and feed ingredients would be generated and implemented in the tool.

The regionalization of emissions models was only implemented for the livestock methane emissions factors by using IPCC guidelines. Easy-to-apply models for European regions are so far not available in order to regionalize other emission models.

The regionalization of the impact assessment method (LCIA) means that different characterization factors are used for each country or for a specific region. The characterization factors of ammonia and nitrogen oxides for terrestrial eutrophication in Romania are higher than the weighted average implemented in SimaPro (Posch et al. 2008). In this assessment a regionalized approach for water depletion was applied (Flury et al. 2012). Furthermore it would be relevant to better differentiate the impacts of different types of land occupation which is not possible with the LCIA method used so far.

Several calculations for direct emissions due to the application of fertilizers and the animal rearing are based on scientific emission models and not on real measurements. One issue of the regionalization is to assess the possibility of having emission models that can be directly fed with data provided by the SME and thus better considering the local circumstances. Some of the models used in this case study are based on regional experiences e.g. in Switzerland. In principle the outcome of these calculations can be influenced by regional circumstances such as rainfall, soil quality, slope of fields, average temperatures, irradiation, etc. Therefore, it would be necessary to better adapt the models to the specific regional situation. But, such easy-to-apply models for the whole of European regions are so far not available. Therefore, only a case specific model for the methane emissions of the animals on the farm was applied according to the tier 2 approach of the IPCC. This emission model is now specific to the Romanian dairy farm (milk yield, animal mature weight, feeding situation etc.). A quite relevant question for a regionalized model would be the calculation of phosphate emissions from erosion and run-off at agriculture areas as well as different type of nitrogen emissions due to fertilizer, manure and dung use. The modelling of NOx emissions from fuel combustion, which depends e.g. on the technology standards applied in a specific region, could be another issue for a regionalized model.

Other aspect of consideration is the sensitivity of different methods for the calculation of the impact categories. For example, climate change or eutrophication is quite robust since the most influential substances are directly related to the primary data. However, other impacts like ecotoxicity or resource depletion are very dependent on the background data, therefore, the differences in the selected database could lead to great differences between LCIA. Regional background data when available should be implemented in the SENSE tool as recommended and regionalized impact assessment and regional emission models which may affect the result will need to be further developed and implemented when available in the future versions of the SENSE tool.

5. Conclusion

The most relevant KEPIs for food supply chains have been selected and implemented in the SENSE web-based tool. The tool is currently being validated by comparing the outcome of the SENSE tool with calculations performed by commercial software (SimaPro and GaBi). The validation is based on using as input only the selected KEPIs. Furthermore, additional case studies where the SENSE tool is tested by users are currently ongoing in the project in at least 30 companies and their supply chains. The harmonized SENSE-tool based on the selected KEPIs is designed to be flexible and adaptable to other food types and will thus be applicable to motivate an LCA based approach and support self-assessment of environmental performance in other food supply chains.

The benefit of a simplified environmental assessment as the SENSE tool can provide will be further assessed in the project by interviews and on-line surveys in companies that will test the SENSE tool.

6. Acknowledgements

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What is the most sustainable biomass supply mix for bioethanol production? Example of the Burgundy region in France

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ABSTRACT

Large uncertainties in the life-cycle greenhouse gas (GHG) emissions of biofuels could offset their potential savings. These arise from the difficulty in taking into consideration land use change effects and N_2O emissions, which are strongly dependent on local pedoclimatic conditions and technological options for feedstock cultivation. Here, we used an crop model (CERES-EGC) to simulate crop growth and C-N dynamics in the cultivation fields at the regional scale for a wide range of lignocellulosic feedstocks. Bioethanol made from cereal straw achieved the largest GHG reduction compared with fossil fuels (at 74%). Among the dedicated energy crops investigated, Miscanthus had the highest GHG abatement potential, ranging from 63 to 72% depending on crop management. A combination of feedstock sources (straw, Miscanthus and triticale) stroke the best compromise to fulfill sustainability criteria, secure the supply and limit the pressure on land and competition with food.

Keywords: lignocellulosic biomass, 2nd generation (2G) bioethanol, emission modeling, Miscanthus, greenhouse gas (GHG)

1. Introduction

Following its ratification of the Kyoto protocol, the European Commission set up directives to promote the development of biofuels in the transport sector, with a 10% target for the share of renewable fuels by 2020. Sustainability criteria were also introduced for conversion units coming onto the market after 2017, to ensure a minimum greenhouse gas (GHG) abatement of 60% compared to fossil equivalents (European Parliament 2009a; European Parliament 2009b). Biofuels are currently produced from agricultural sources. The agricultural sector, however, represents 15% of French GHG emissions, and 89% of N_2O emissions (CITEPA 2013). N_2O shows a high global warming potential (GWP), 298 higher than CO_2 (IPCC 2007). It is thus essential to accurately estimate GHG emissions from biomass crops to supply bioethanol conversion units.

Life cycle assessment (LCA) is an objective and holistic tool, commonly used to estimate the environmental impacts of biofuels. Whereas the uncertainty band in the GHG balance of fossil fuels typically amounts to 4 g CO_2 eq MJ^{-1}, it can be 2.5 to 10 times higher for first generation (1G) biofuels (10 to 40 g CO_2 eq MJ^{-1}) (Edwards et al. 2011) and not estimated for second generation (2G) biofuels due to the lack of data. Even if guidelines have been proposed (ADEME 2010), the methodology for GHG accounting lacks consensus on two crucial aspects: N_2O emissions and land use change (LUC) effects. Both of them could offset the conclusions on GHG savings (Searchinger et al. 2008; Hoefnagels et al. 2010; Cherubini and Strømman 2011; Smith and Searchinger 2012). In most cases, LUC is ignored or minimized, but recent studies encourage the integration of this factor within the biofuel sustainability criterion (ADEME 2012; De Cara et al. 2012; European Parliament 2012). N_2O emissions are mostly calculated from IPCC generic factors without considering any local pedoclimatic variations, which generates large uncertainties (Hoefnagels et al. 2010; Smith and Searchinger 2012).

As shown in recent publications (Dufossé et al. 2013), crop modelling can overcome these problems by a better consideration of local soil and climate variations and their effects on yield and GHG emissions. Combined with scenarios of bioenergy crop establishment, crop modelling makes it possible to define multi-sources feedstock supply scenarios and minimize GHG emissions from biofuel feedstocks. Whereas it is now obvious that the GHG abatement potential of 1G biofuels is limited (ADEME 2010; ADEME 2012; Humpenöder et al. 2013), 2G biofuels are still under technological development and their feedstock (lignocellulosic biomass) remains to be fully evaluated. In particular, different types of feedstock have to be compared to define the most reliable and sustainable mix for a given conversion unit.

The aim of this paper is thus to evaluate feedstock supply scenarios, through a LCA, for a unit of production of 2G bioethanol, considering field emissions, agricultural operations, agricultural input (fertilizers, pesticides, herbicides) manufacturing, as well as input and biomass transportation.

2. Methods

2.1. Crop modeling

Regional modeling was based on the crop model CERES-EGC (Gabrielle et al. 2002). The model requires meteorological and management data as forcing variables, as well as soil and vegetation data as input factors and runs at a daily time step. CERES-EGC comprises a physical sub-model which simulates the transfer of heat, water and nitrate down the soil profile, as well as soil evaporation, plant water uptake and transpiration in relation to climatic demand (Gabrielle et al. 2002). A biological sub-model simulates the growth and phenology of the crops and a microbiological sub-model simulates the turnover of organic matter in the ploughed layer. Direct field emissions of CO_2, N_2O, NO and NH_3 into the atmosphere are simulated with different trace gas modules (Lehuger et al. 2009; Lehuger et al. 2010). The nitrous oxide emission module simulates the production of N_2O in soils through both the nitrification and denitrification pathways. N_2O emissions resulting from both processes are soil-specific and are proportions of total denitrification and nitrification (Lehuger et al. 2009). CO_2 exchanges between the soil-plant system and the atmosphere are modeled from net photosynthesis and soil organic carbon (SOC) mineralization (Lehuger et al. 2010).

Soil input factors include physical properties, soil texture characteristics and biological parameters for nitrification and denitrification processes. At regional scales, these inputs are inferred from the 1:1 000 000 soil map of the European soil map, in which the twenty soil classes occurring in France were reduced to fourteen main soil types after an aggregation based on their characteristics (Dufossé et al., 2013). The model was run on simulation units (SU) defined by overlaying the soil map with geo-referenced databases on meteorology, land cover, administrative borders and crop management, as detailed in Dufossé et al. (2013). The area of the SU varied from 2.5 ha to 30 000 ha, with a median value of 1 210 ha. Administrative borders of regions and departments were given by IGN-GEOFLA and InfoSIG Cartographie (2010) and were used to determine crop management by deriving regional statistics (Agreste 2008). Corine Land Cover dataset (European Environment Agency 2006) provided the detailed area of utilizable agricultural land (UAL) within each SU, broken down into arable lands, fragmented lands and grassland areas. Meteorological datasets were provided on a 8km-mesh (SAFRAN grid, (Pagé et al. 2009)), each SU being associated to the closest grid point. The results presented in this paper at regional supply-area levels resulted from the weighted means of individual results from the SU involved.

2.2. Study domain and system boundaries

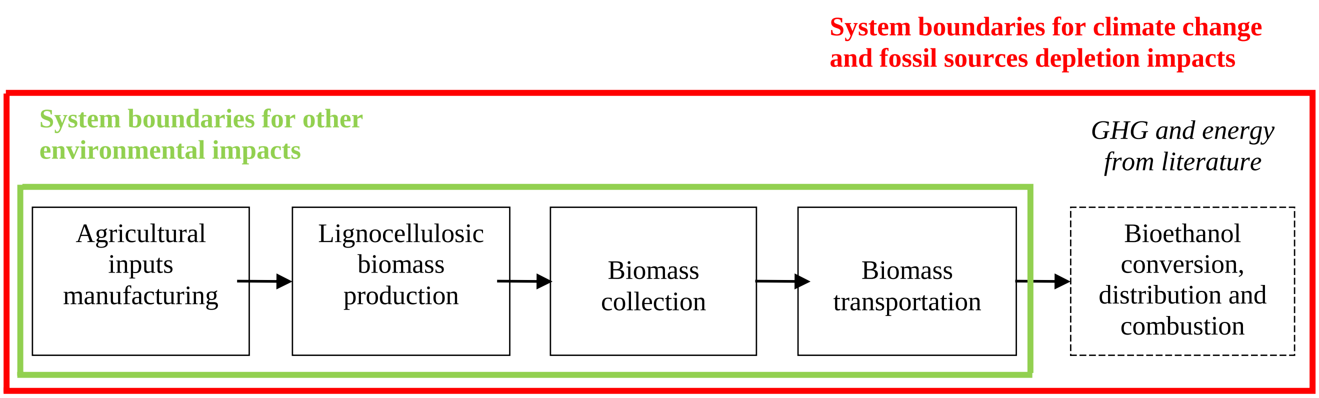

Figure 1: System description and boundaries for climate change and fossil resources depletion impacts (red) and other environmental impacts (green)

The LCA methodology applied in this study followed the ISO 14044 standard requirements. The biomass supplied to the bioethanol conversion unit represents the downstream boundary (green boundaries, Figure 1). The system was extended to the production of bioethanol to allow a comparison with a reference fossil fuel, in terms of GHG and energy balances. Data of the biomass conversion process were extracted for the theoretical unit presented by the National Renewable Energy Laboratory (Aden et al. 2002), for which emissions and consumptions were evaluated by a later study (Edwards et al. 2013) (Figure 1).

This unit can produce 600 t ethanol d^{-1} [annual production of 219 000 t ethanol], from a biomass supply of 730 000 t DM yr^{-1} [i.e. with a process mass efficiency of 30% kg EtOH kg $biomass^{-1}$]. These estimations were based on a single biomass source of woody short rotation coppice (SRC). Because of the lack of adequate data in the literature, we assumed this conversion efficiency to be independent of the type of input biomass.

The Burgundy region (Northeastern France) was thus selected as representative of the agricultural situation of France, presenting in particular a supply area large enough to supply the bioethanol conversion unit. Besides, a simulation period of 20 years appeared to be a minimum to capture inter-annual climatic variations and take into account a whole life-cycle of perennial crops. For these reasons, the 2010-2030 time slice was selected for the climate data. All results presented in this paper are thus annual means from the whole 20-year life cycle. Several biomass sources were studied: crop residues (cereal straw) and dedicated crops, either annual (sorghum and triticale) or perennial (Miscanthus). A feedstock composed from a mix of biomass sources, based on wheat straw, Miscanthus and triticale, was also examined.

2.3. Life cycle assessment (LCA)

2.3.1. Methodological aspects

Table 1 sums up the methodological choices made for the LCA study. N_2O emissions estimates LUC effects are more detailed within the life cycle inventory section below. As infrastructures represent a t

Table 1. Summary of methodological choices for the LCA

Methodology aspect	**Choice**	**Reference**
Functional unit	Feedstock supply at the conversion unit entrance (730 kt DM yr^{-1}) and 1 MJ of producted and used biofuel for GHG and energy comparison	(European Parliament 2009a)
LCA software and database	Simapro 7.3.3 and Ecoinvent 2.2	(Frischknecht et al. 2007; PRé Consultants et al. 2010)
Allocation rules between products and byproducts	Energetic prorate applied in the NREL process calculations, from biomass to bioethanol No byproducts were considered in the biomass production	(European Parliament 2009a)
GHG emissions calculations: - Biogenic carbon - Emissions factors - N_2O emissions	- GWP (biogenic CO_2) = 0 and GWP (fossil CO_2) = 1 - IPCC 2007 GWP considered for 100 years - Direct N_2O emissions from CERES-EGC model outputs - Indirect N_2O emissions from CERES-EGC model outputs (NH_3, NOx and NO_3^-) combined with IPCC 2006 equations	(IPCC 2006; IPCC 2007)
Land use changes (LUC)	Both direct and indirect LUC were considered: - Direct LUC estimated from CERES-EGC model outputs and scenarios - Indirect LUC from literature	(European Parliament 2009a; ADEME 2012; De Cara et al. 2012; European Parliament 2012)
Infrastructures consideration	Infrastructures were neglected (roads, buildings)	(European Parliament 2009a; ADEME 2010)

2.3.2. Life cycle impact assessment (LCIA)

The IMPACT 2002+ (Jolliet et al. 2003) method was selected, since it allows to evaluate midpoint and end-point impacts and also since it is based on the IMPACT 2002 model (Pennington et al. 2005), the Ecoindicateur 99 method (Goedkoop and Spriensma 1999) and the CML method (Guinée et al. 2002). As midpoint impacts are estimated from substance equivalences, they present fewer uncertainties than end-point impacts, based on models and strong hypotheses. The results presented in this study are therefore derived mainly from mid-point impacts.

A particular effort was carried on GHG balance and climate change impact to evaluate the sustainability of biofuels. The time horizon in IMPACT 2002+ is set to 500 years to consider long term impacts, especially for CO_2 that contributes to climate change for such a period. However, a time horizon of 100 years for the substances impacting the greenhouse effect is usually considered into the other methods, even if it does not reflect the whole impacts caused by these substances. We chose a time horizon of 100 years in this study.

In order to consider the energy efficiency of produced biofuels, the non-renewable energy consumption has to be estimated. Finally, since these fuels are based on biomass production, impacts specifically linked with agricultural production and affecting the ecosystems were considered, such as terrestrial and aquatic acidification, ecotoxicity, aquatic eutrophication and land occupation.

GHG balance uncertainties were estimated from modelling outputs (yield, N_2O, NO_x, NH_3 and NO_3^-) as N_2O emissions have the highest weights in biofuel GHG balance (Cherubini and Strømman 2011), excluding uncertainties from biomass conversion to bioethanol. They were calculated for individual SU from standard error of the annual results on a 20-year period (2010-2030), and then aggregated to region scale weighted by area.

2.4. Life cycle inventory (LCI)

2.4.1. Cropping management

- Agricultural residues

A reference crop succession was determined in Burgundy from statistical data and experts' knowledge, as follows: winter barley – oil seed rape – winter wheat (Agreste 2008; Lesur 2012). The cropping management of reference crops was determined through regional studies (Agreste 2008) with a unique level of fertilization for the whole region. Straw exportation rates varied from 33% (one removal every three years) to 50% and were determined for each soil type and crop succession to maintain soil C stocks (FRCA Picardie et al. 2009), by assuming that the rates determined in Picardy were applicable in Burgundy. Wheat straw was favored due to its high yield, and sometimes, completed by barley straw. Livestock requirements, estimated from literature (FCBA and PNRB 2009), were first subtracted to the straw thus produced, before assessing the amounts of straw available for bioenergy.

- Dedicated annual and perennial crops

As sorghum and triticale are annual crops, they can easily be inserted in crop succession, and thus be grown on any arable area within the region. Barley was thus substituted by the annual energy crops within the succession. Sorghum and triticale crops were silage-harvested (with a biomass at 70 and 60% of moisture respectively) while both early (autumn) and late (spring) harvests were simulated for Miscanthus (RMT Biomasse 2013). An organic variant for the management of this crop, without chemical inputs, was also implemented ("Misc.-0input", Table 2). In order to minimize the inputs, crop management was adapted to estimate crop yields and the resulting nutrient export rates (Table 2) (Cadoux et al. 2012; Béjot 2013). Early harvests ("Misc.-early") lead to higher yields with higher moisture content (around 60%), whereas late harvests ("Misc.-late") increase nutrient recycling within the rhizomes, while decreasing fertilizer requirements (Table 2) and the moisture content of harvested biomass (15%). As no data was available on the removal phase of Miscanthus, experimental measurements were used to apply the LCA on GHG emissions and measurement uncertainties were added in the miscanthus GHG balance (Dufossé et al. 2014).

- Mix of biomass sources

To provide an alternative perspective, a feedstock supply mix composed of three biomass sources in equal proportions (243 kt DM yr^{-1}) was assessed. It comprised cereal straw, triticale (which came out as the best annual energy crop in this study) and Miscanthus. In order to approach real establishment of energy crops, hypotheses on area restriction were carried Miscanthus and triticale. No hypothesis on area restriction was carried for straw. Miscanthus was assumed to be planted exclusively on marginal lands, as set aside land, to avoid competition with food crops. Protected areas were excluded. A late harvest and adapted fertilization rates were applied to Miscanthus ("Mix-Misc.", Table 2). Triticale was assumed to be on the less productive arable lands to minimize competition with food crops ("Mix-triti.").

Table 2. Mean fertilization rates applied to the crops for simulations. Mean annual rates, weighted by arable area within the SU. Three management for Miscanthus were simulated: early harvest with fertilization ("Misc.-early"), a late harvest with fertilization ("Misc.-late") and a late harvest without fertilization ("Misc.-0input"). Two annual crops were also simulated ("Sorghum" and "Triticale"), as well as a mix of biomass composed of a third of Miscanthus with fertilization and late harvest on set aside lands ("Mix-Misc.") and a third of triticale on low productive land ("Mix-triti.").

	Misc.-0input	Misc.-late	Misc.-early	Sorghum	Triticale	Mix-Misc.	Mix-triti.
N fertilization (kg N ha^{-1})	0	54	120	144	145	49	110
P fertilization (kg P ha^{-1})	0	7	14	24	25	6	18
K fertilization (kg K ha^{-1})	0	83	160	200	140	78	106

2.4.2. Direct and indirect emissions

Emissions of nitrogen compounds (N_2O, NOx, NH_3 and NO_3^-) in soil, water or atmosphere were simulated from the CERES-EGC model (Table 3). Indirect N_2O emissions were estimated from IPCC emission factors applied on simulated emissions of NOx, NH_3 and NO_3^- (IPCC 2006). Heavy metals inputs related to fertilization, and their transfer to soils were estimated from fertilization rates and emission factors proposed by Nemecek and Kägi (2007). Because of the lack of reliable data on heavy metals inputs through pesticides and exports by biomass, these fluxes were ignored.

Table 3. Main outputs of the CERES-EGC simulations, as inputs of the LCA. Mean annual rates, weighted by arable area within the SU.

	Misc.-0input	Misc.-late	Misc.-early	Sorghum	Triticale	Mix-Misc.	Mix-triti.
Yield (t DM ha^{-1})	16.13	16.69	22.56	13.46	13.73	15.6	9.0
Direct N_2O (kg N ha^{-1})	0.27	0.43	0.55	1.57	0.97	0.45	0.17
NO_3^- (kg N ha^{-1})	12.55	11.46	15.18	45.90	40.42	13.15	78.70
NH_3 (kg N ha^{-1})	-0.14	0.12	3.15	3.44	0.99	-0.01	0.16
NOx (kg N ha^{-1})	0.59	0.58	0.81	0.76	0.64	0.54	0.61
Net soil C variation (t C ha^{-1})	0.54	0.57	0.65	-1.11	0.19	0.91	0.07

2.4.3. Transportation

Because the lack of information on transportation, especially for the Burgundy region, the following hypotheses for distances were applied: 25 km between the farm and the regional storage where agricultural inputs (seeds, fertilizers, pesticides, fungicides and herbicides) are stored, and 100 km between fields and the region where the biomass is stored for the conversion unit supply, as in the JRC/EUCAR/CONCAWE study for European biomass (Edwards et al. 2013). All vehicles were selected among the most efficient vehicles in the Ecoinvent database (complying with the EURO5 standard, Frischknecht et al. (2007)).

2.4.4. Land use changes

Direct and indirect LUC (dLUC and iLUC, respectively) were considered separately, in a second part of the study. Firstly, dLUC from land conversion were estimated from variations in soil C stocks caused by the integration of bioenergy crops within the crop succession. These soil C stock variations were estimated from modeling outputs. Then, variations in soil C stock from the reference crop succession were withdrawn to obtain net stock differences (Table 3). These differences were transformed into CO_2 emissions according to the IPCC equations (IPCC 2006). For cereal straw, the dLUC was neglected since the exportation rates applied for modeling allowed to maintain soil C stocks (FRCA Picardie et al. 2009). To estimate the dLUC effects of converting set aside to cropland, the soil C variations of the reference crop succession were replaced by those estimated by simulating a set aside land with CERES-EGC before been subtracted to the soil C variations of energy crop succession.

Indirect LUC effects were mostly related to the displacement of food crops, because of the implantation of bioenergy crops. They were estimated at global scale, based on a recent meta-analysis (De Cara et al. 2012). For annual crops (sorghum and triticale), the CO_2 emissions due to iLUC were assessed at 37 g CO_2 eq MJ^{-1} (Al-Riffai et al. 2010; De Cara et al. 2012), whereas they were assessed at 27 g CO_2 eq MJ^{-1} for Miscanthus. These values correspond to the top of the range of values for every type of biomass from the meta-analysis. In scenarios where crops were established on set aside lands, no CO_2 emissions from iLUC were considered.

2.4.5. Biomass conversion into bioethanol

Emissions for the conversion of lignocellulose to ethanol were taken from the 2G bioethanol conversion unit described in the JRC/EUCAR/CONCAWE study ('WW/WFET1' scenario), as 15.6 g CO_2 eq MJ^{-1} including also conditioning and bioethanol distribution (Edwards et al. 2013).

This 2G bioethanol conversion unit also consumes 1.97 MJ of energy to produce 1 MJ of bioethanol (1.97 MJ $MJ(EtOH)^{-1}$). This consumption was split into multiple sources: non-renewable energy sources (0.28 MJ $MJ(EtOH)^{-1}$), nuclear power (0.01 MJ $MJ(EtOH)^{-1}$), biomass itself and heat recycling within the process (Edwards et al. 2013). In this study, for an equivalent conversion process, the energy consumption related to feedstock supply (poplar SRC) was estimated at 0.11 MJ $MJ(EtOH)^{-1}$. As non-renewable energy consumptions were not detailed according to bioethanol production steps, we assumed that the energy used in the agricultural phase was only non-renewable energy. For the bioethanol conversion phase, the non-renewable energy consumption was thus assumed at 0.18 MJ $MJ(EtOH)^{-1}$.

3. Results and discussion

3.1. GHG balance of bioethanol production

3.1.1. Global GHG balance

Figure 2 shows the GHG balance of bioethanol production from different lignocellulosic biomass sources. Thanks to the integration of local pedoclimatic variations in modeling crop production, the uncertainties on biofuel LCA results decreased from 1 to 4 g CO_2 eq MJ^{-1} for biomass production, excluding uncertainties from biomass conversion to bioethanol. The uncertainties were thus reduced ten-fold compared to recent studies (Edwards et al. 2011), improving the accuracy of estimates of GHG savings for 2G biofuels.

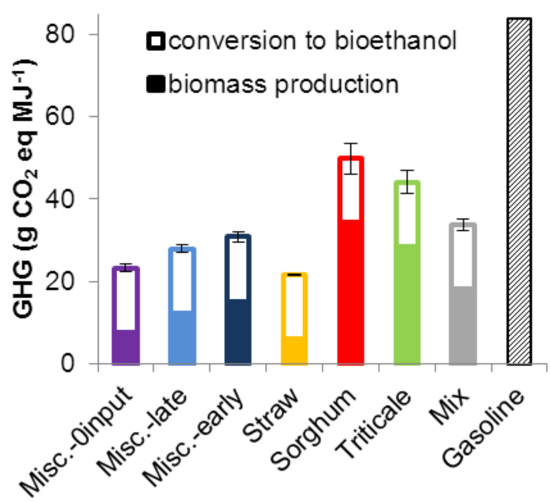

Figure 2. GHG balance of bioethanol produced from crop residues (straw), annual crops (sorghum and triticale), perennial crops (Miscanthus) and the mix of three biomass sources (straw, triticale and Miscanthus). GHG emissions of gasoline is also presented as the fossil reference. Error bars show the standard deviation of errors due to spatial and temporal variability within the region.

In terms of feedstock, Miscanthus (whatever its management) or cereal straw achieved larger GHG savings than annual crops, both reaching the 60% GHG threshold set by the RED Directive for 2017 (European Parliament 2009a).

The production of Miscanthus without fertilizer inputs (Misc.-0input) emitted 7.7 g CO_2 eq MJ^{-1}, which is similar to the 7.2 g CO_2 eq MJ^{-1} estimated as GHG emissions for poplar SRC cropping (with 25 kg N ha^{-1}) within JRC/EUCAR/CONCAWE study (Edwards et al. 2013). Both other Miscanthus management (Misc.-late and Misc.-early) received fertilization and emitted twice GHG than the one without inputs (12.4 and 15.2 g CO_2 eq MJ^{-1}). However, the required amount of area converted to Miscanthus (2.5% of UAL) seems unrealistic to supply the conversion unit with Miscanthus as a unique source of 2G bioethanol in the near future.

Since straw was considered as a crop residue, all impacts from wheat growing were attributed to grain production. Therefore, only impacts for biomass collection were attributed to straw, which minimized its GHG balance. In the JRC/EUCAR/CONCAWE study (‘STET1’ scenario), GHG emissions from straw were estimated to 3.8 g CO_2 eq MJ^{-1}, which is in the same order of magnitude as our findings (6.0 g CO_2 eq. MJ^{-1}). Since the study did not consider livestock in the estimation of available straw, yield per unit area should have been higher (in our study: 0.8 t DM ha^{-1} yr^{-1}) and therefore emissions due to collection, lower. Besides, due to these high livestock requirements (748 kt DM yr^{-1}), the amount of available straw could only supply 50% of the biomass required by the conversion unit (372 kt DM yr^{-1} produced for 730 kt DM yr^{-1} required).

Annual crops showed mediocre results when combining their high fertilizer requirements, high GHG emissions and moderate yields (around 12.5 t DM ha^{-1} yr^{-1}) in Burgundy. However, their introduction into crop successions could suffice to fulfill the biomass requirements of the conversion unit. A mix of biomass sources (residues, annual and perennial) is likely to be an interesting compromise to supply the conversion unit by ensuring feedstock supply while maximizing GHG savings (60% of GHG reduction for fossil reference, Figure 3).

Field N_2O emissions were the main contributor to GHG emissions for all biomass sources excluding Misc-0input, followed by emissions due to biomass transportation and fertilizer production (data not shown). Therefore, changes in land use have to be carefully thought to optimize biomass performances, by selecting fields which produce high yields and minimize N_2O emissions while located close to the conversion unit. Selecting annual crop variety with low input requirements is also a worthy option for improving biomass GHG balance, as also recommended by Smith and Searchinger (2012).

3.1.2. Effects of land-use changes (LUC) on GHG balance

The results presented above did not consider the effect of LUC on the GHG balance of feedstocks and bioethanol. Direct LUC effects were positive for Miscanthus (Figure 3b), as Miscanthus stored large amounts of C in soils. GHG savings from Miscanthus-based ethanol were between 63 and 72% without LUC effects and between 79 and 90% with them. However, due to significant and negative dLUC effects, sorghum GHG savings became negative (-4%), which means that, in Burgundy, producing and using bioethanol from sorghum results in higher GHG emissions compared to the reference fossil fuel.

When considering dLUC and iLUC, only straw biomass ensured biofuel GHG savings larger than 60% (Figure 3c). Miscanthus managed without inputs and the mix of biomass presented interesting results (58 and 57% of GHG savings respectively). Misc-0input had an overall acceptable GHG balance thanks to low N_2O emissions and the absence of fertilizer inputs. The mix of biomass sources compensated the poor GHG performances of triticale (11% of GHG savings) by the excellent ones of straw (74% of savings) and the advantageous ones of Miscanthus: no inputs (minimizing GHG emissions), high soil C storage (positive dLUC effects) and implantation on set aside lands (no iLUC effects).

The iLUC values used in this study came from the literature. They were generic and taken in their upper range. Due to the weight of the iLUC effects, further studies would be necessary to yield conservative estimates of GHG savings from 2G biofuels. Marginal lands showed reliable potentials for bioenergy cropping since the biomass produced would not compete with food production, and generate little iLUC effect.

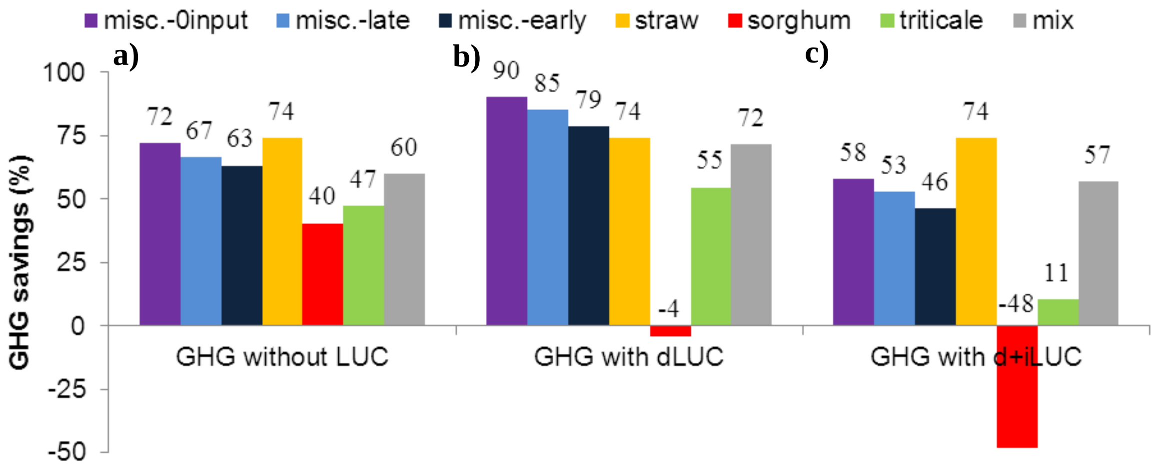

Figure 3. GHG balance of biofuels produced from different biomass sources, without considering LUC effects (a), with the integration of direct (dLUC, b) and indirect LUC (d+iLUC, c) effects in Burgundy.

3.2. Energy balance

Figure 4 showed the non-renewable energy consumptions for bioethanol production from different lignocellulosic biomass sources. Compared to the fossil reference, energy savings ranged from 65% for sorghum to 78% for Miscanthus without chemical inputs. These savings were highly dependent of the amount of cropping operations and chemical inputs. Biomass yields also played a role for these estimates as straw required few field operations but also produced low mean yields per hectare. Even if the conversion process was assumed to be similar for all studied biomass sources, the energy efficiency of 2G bioethanol was high in all cases and likely to increase as industrial processes are improved.

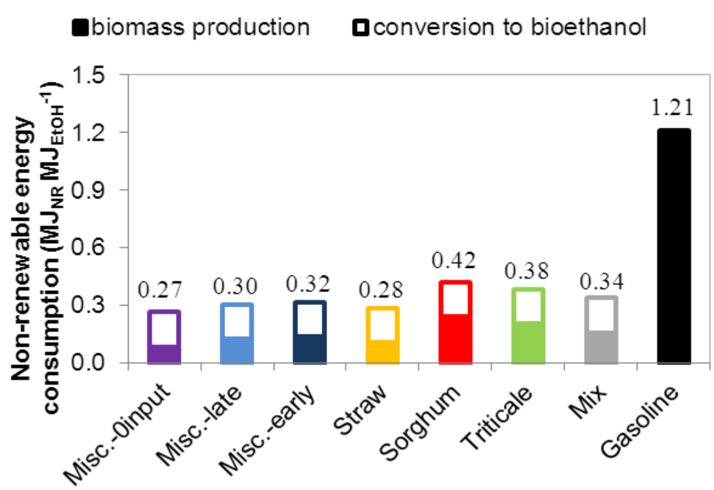

Figure 4. Non-renewable energy consumptions of 2G bioethanol produced from different biomass sources in Burgundy and comparison with fossil reference.

3.3. Other environmental impacts

Since few details were available on the conversion process for 2G bioethanol, the assessment of other environmental impacts focused exclusively on the biomass production to compare the various feedstocks. Crop receiving higher chemical inputs (herbicides, fungicides, fertilizers and associated heavy metal inputs) showed larger impacts on the ecosystem (scores of 63 to 100% of maximum impacts for eutrophication, aquatic and terrestrial acidification and ecotoxicity). Land occupation was directly related to crop yields. Sorghum and triticale were thus more harmful for the ecosystem (Figure 5). Compared to its other management, Miscanthus with early harvest turned out to affect notably the ecosystem, even if land occupation impact was lessened (Figure 5) thanks to high yields (averaging 23 t DM ha^{-1} yr^{-1}, Table 3).

Within the unique score assessed with the IMPACT2002+ method, damages on ecosystem had the bigger share compared to all endpoint impacts (data not shown). Since these impacts are strongly related to fertilizer input rates and in order to reduce the environmental impacts of bioethanol, it is crucial to select biomass sources that require the lowest chemical inputs, such as Miscanthus or agricultural residues. It is also possible to mitigate impacts if choosing a feedstock supply composed from a mix of biomass sources with straw, Miscanthus and triticale in equal shares. The considerable impacts of the last one (scores between 63 to 100% of maximum impacts) were mitigated by the limited impacts of Miscanthus and mainly straw to moderate impacts (scores of 35 to 79%).

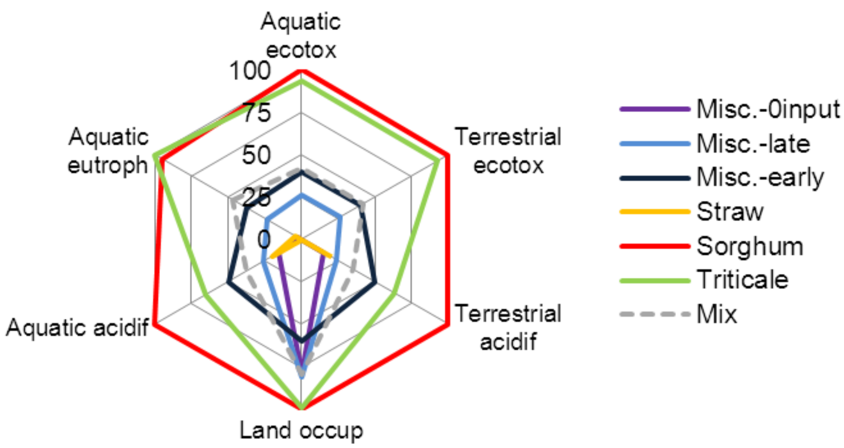

Figure 5. Results of midpoint impact assessment, of the damages on ecosystem endpoint impact, for different biomass production in Burgundy. Impacts are presented as a proportion of the highest score in the midpoint impact category.

4. Conclusion

Using a crop model to integrate the local pedoclimatic variability within a supply area made it possible to provide more accurately GHG balances for a conversion unit of 2G bioethanol from different sources of biomass. The estimated GHG savings, obtained with the LCA thus produced, could help for biomass selection and ensure the potential GHG reduction of 2G bioethanol from agricultural residues and Miscanthus. These bioethanol chains could fulfill the sustainability criteria set by the European Parliament. Sorghum and triticale, used as unique feedstock source, showed the lowest GHG savings. Nonetheless, if they are integrated in a feedstock mix with straw and Miscanthus, they can contribute to ensure the biomass requirements and limit biomass storage at the conversion unit, while their mediocre environmental performances could be mitigated by the other sources of biomass.

As shown by the previous results, the integration of LUC effects (direct or indirect) could offset the conclusions on GHG savings. When dLUC and iLUC were included in the GHG balance, only straw could ensure GHG savings higher than 60%. Since amounts of cereal straws are limited in the studied area, other biomass sources have to be considered. It is thus essential to accurately evaluate emissions, especially due to iLUC, and guide the establishment of energy crops towards zones that minimize emissions, such as set aside or marginal lands with low productivity in food cropping. However, these low productive lands may also show mediocre environmental and yield performances for energy crops, especially for cereals such as triticale.

Bioethanol from all energy crops presented in this study showed non-renewable energy consumptions reductions higher than 65% compared to fossil fuels. Best performances were obtained by straw and Miscanthus. The main environmental impacts of these crops on ecosystems were directly linked to the use of chemical inputs (fertilizers or herbicides). Biomass sources requiring the lowest chemical inputs rates should be favored in order to limit environmental impacts.

The study of innovative energy crops, such as Miscanthus, switchgrass or SRC, is likely to be critical to the optimization of biomass feedstock. Before widening the portfolio of biomass feedstocks for 2G bioethanol, we suggest using a mix of biomass sources for feedstock supply, made up of highly performing crops (Miscanthus), residues (straw) and commonly produced agricultural crops (triticale). Also, a particular interest should be paid to crop establishment patterns, and to the location of the biofuel conversion unit. Maps of potential yields, N_2O emissions and soil C sequestration rates, such as generated here with the ecosystem model, would be very useful and practical tools to guide public and private stakeholders in this direction.

5. Acknowledgments

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Agri-Footprint; a Life Cycle Inventory database covering food and feed production and processing

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ABSTRACT

Agri-footprint is a new life cycle inventory database that focuses on the agriculture and food sector. The goal of this database is to support life cycle assessment practitioners to perform high quality assessments. The database contains a methodologically consistent dataset for a large number of crops, crop products, animal systems and animal products. These inventories can be used as secondary data in LCAs. Non-LCA models were used to calculate a wide array of elementary flows (such as land use change, water use, fertilizer application rates), to support assessment on a multitude of environmental issues. To safeguard relevance and data quality, the database will be updated regularly. As the public interest in food LCAs is expected to increase in the near future, Agri-footprint will be a helpful resource for practitioners in this field.

Keywords: Life cycle inventories, food, agriculture, food processing, crop cultivation

1. Introduction

The goal of the LCI database Agri-footprint is to facilitate life cycle assessments (LCAs) in the domain of food and agriculture by making life cycle inventory (LCI) data available in a single, internally consistent database. LCA studies of food have become more frequent in the last few years, although studies of food and agriculture have been undertaken almost since the emergence of the LCA method. The first agricultural LCAs were performed in the early 1990s. Pioneering food LCA case studies include LCAs about tomatoes, by Gysi and Reist (1990), ice cream, by Bolliger and Zumbrunn (1991) and Margarines, by Unilever (Vis, Krozer, van Duyse, & Koudijs, 1992). The proportion of published agriculture-related LCAs in the international Journal of LCA increased considerably at the end of the 2000s and has remained around 25% since then (Figure 1). Some reasons for this increased share are:

- Public interest in the first edition of the FAO report ‘Livestock’s Long Shadow’ (Steinfeld et al., 2006) which put the environmental impact of agriculture and especially animal production on the agenda.
- Pressure from retailers to provide carbon footprint information on food products since around 2007.
- The introduction of European biofuel legislation in 2009 (European Commission, 2009) and its related policy issues on fuel, feed and food competition.
- Specific agricultural related impact categories and related methodological solutions came up, such as water resource depletion and land use change.

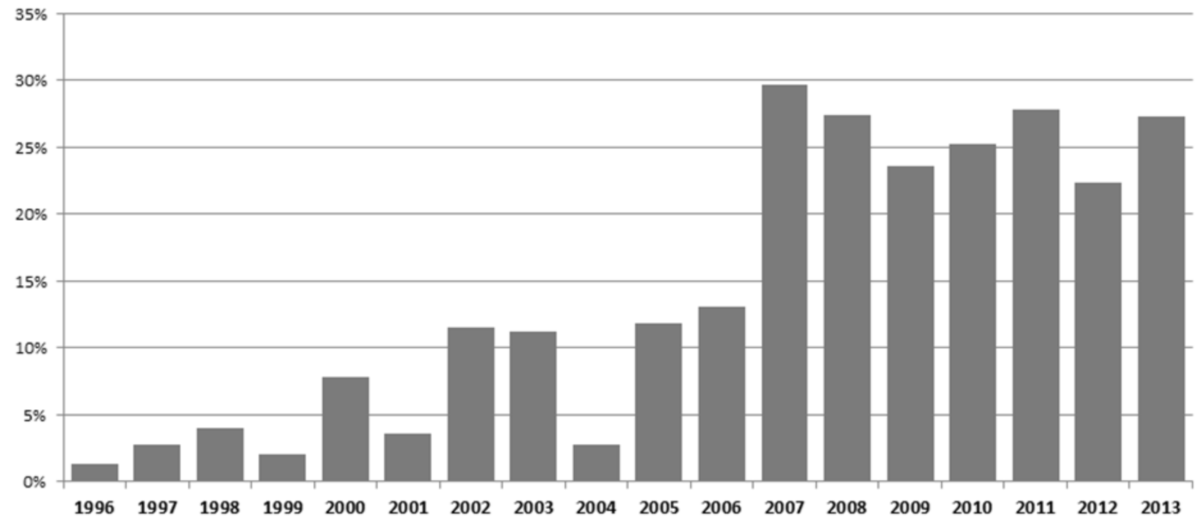

Figure 1. Share of agricultural related LCAs in the International Journal of LCA. Based on our survey of published articles.

In coming years, many food related LCAs will be performed due to the special attention from the European Commission for food, feed and beverages in the Product Environmental Footprint (PEF) program. Also the European research and innovation program Horizon 2020 focuses on more sustainable food production systems which have to include a sustainability assessment in line with the ILCD handbooks.

Therefore there will also be more and more demand for reliable and consistent secondary data in this field. The aim of Agri-footprint is to fulfill this demand by bringing data and methodology together and make it easily available for the LCA community. By having these generic LCIs readily available, future LCAs can be conducted more efficiently and also more reliable.

Although there are already a number of extensive LCA databases available (such as Ecoinvent, (2013) and ELCD (JRC-IES, 2012)), agricultural (processing) and food production LCIs only represent a small fraction of the data covered. Relevant information from a large number of food LCIs and food statistics (e.g. from FAO) are not yet implemented in LCA databases.

In addition, agricultural and food production companies find it often difficult to find a practical way to share their environmental information with the LCA community. Agri-footprint aims to combine information from these disparate sources (scientific literature, statistics and company specific data), into a single internally consistent LCI database. Therefore, Agri-footprint will also serve as a platform for companies to communicate their specific LCI data to a relevant audience. Already, some company specific life cycle inventories are included in Agri-footprint (currently Nutramon fertilizer from OCI and sugar products from Suiker Unie). By making the better performance of specific companies visible, LCA users can more easily identify improvement options in a lifecycle assessment.

The release date of the first version of the database is spring 2014. The database is intended to be used in the public domain and is available to LCA and sustainability experts who have a SimaPro license.

Agri-footprint is intended to be used as a secondary data source or background data for another LCI or LCA (comparative/ non-comparative). More specifically, potential applications of Agri-footprint may be:

- the identification of key environmental performance indicators of a product group
- hotspot analysis of a specific agricultural product.
- benchmarking of specific products against a product group average.
- to provide policy information by basket-of-product type studies or identifying product groups with the largest environmental impact in a certain context.

2. Methods

Agri-footprint includes linked unit process inventories of crop cultivation, crop processing, animal production systems and processing of animal products for multi-impact life cycle assessments. Agri-footprint also contains inventory data on transport, fertilizers production and auxiliary materials. Agri-footprint uses some background data that was sourced from ELCD (JRC-IES, 2012) and USLCI (National Renewable Energy Laboratory, 2012) datasets. Detailed reports are available for the methods and guiding principles (Blonk Agri-footprint BV, 2014a) and the sources and treatment of data (Blonk Agri-footprint BV, 2014b).

2.1. Guiding principles

During the development of Agri-footprint, the first step was to create data that was of consistent quality for all crops and regions covered. For example, all fertilizer application rates, fertilizer types, water use are based on the same methodologies for all crops. To create this consistent baseline dataset, data were derived from documented expert data or data from statistics.

Agri-footprint contains attributional LCIs of crop production, crop processing, animals systems, transport and other background processes (see Figure 2 as an example of types of processes included in database). Generally average mixes are considered that are representative for the specific product and location.

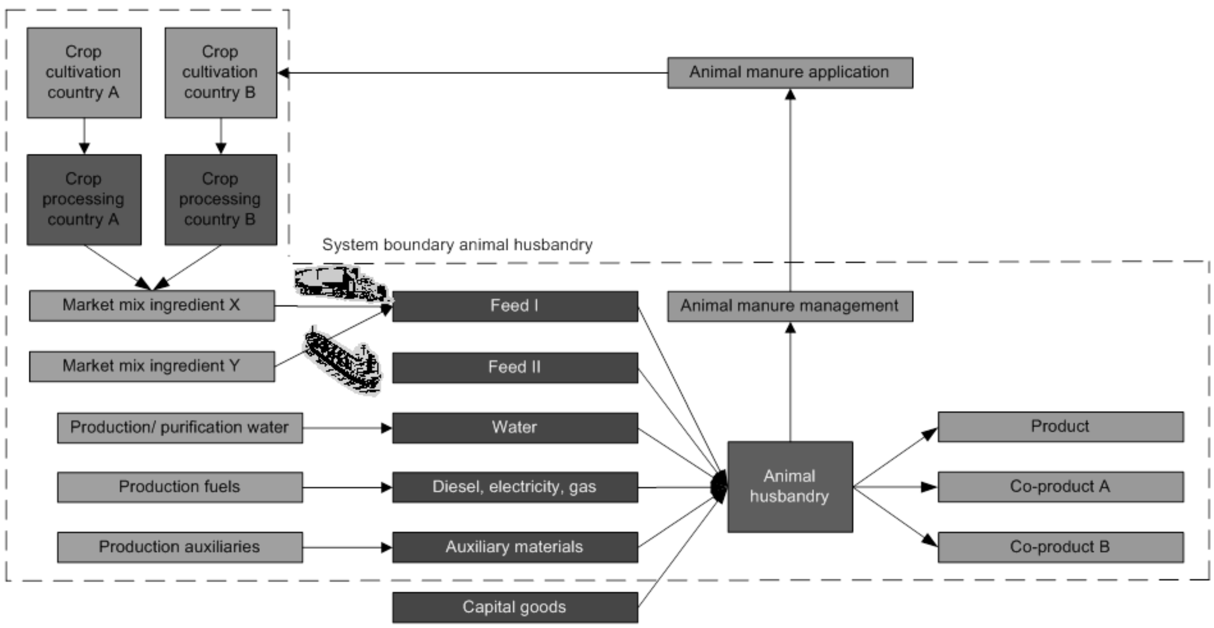

Figure 2. System boundaries of an Animal husbandry LCI in Agri-footprint.

The main baseline data source is the public domain (scientific literature, statistics from (FAOstat, Eurostat, IFA, etc.), using rolling 5 year averages. Data from the public domain are assessed based on representativeness (time-related coverage, technical coverage and geographical coverage), completeness, consistency and reproducibility. When data from public or confidential research by Blonk Consultants are more representative, complete and consistent, these data are used. Where possible, the data have been reviewed by industry experts and, where fit, an uncertainty range and a distribution type has been attributed.

The data in Agri-footprint are derived from different sources. The LCIs for animal husbandry, transport, auxiliary materials, fertilizers etc. have been developed based on previous reports/studies by Blonk Consultants.

Fertilizers production was modeled based on the latest available literature and a specific fertilizer product was based on primary data from a large Dutch fertilizer producer (Calcium Ammonium Nitrate produced by OCI Nitrogen in the Netherlands). Auxiliary materials were based on the ELCD 3 database or literature sources. For some (deep) background processes, estimates had to be made (e.g. the production of asbestos which is used in sodium hydroxide production which is used in vegetable oil refining), and these processes are of lower quality and representativeness.

Processing inventories were drawn from the feedprint study (Vellinga et al., 2013). These inventories are generic for all provided countries and regions. These processes are either largely similar between countries or the data available was not specific enough to create country/ region specific processes. These generic processes are regionalized by adapting the inputs for energy consumption to the country or region where the processing takes place. This means that the processing (in terms of mass balances, and types of inputs) is generally the same for all regions, except the input processes for energy consumption are specific per country. This means that the representativeness may have decreased for these processes. Transport distances and modes from and to the processing plant are also country specific. The geographical representativeness will be improved in future upgrades of Agri-footprint.

The aim is for the LCI data to be as recent as possible, which means that when better quality data or statistics on the processes/ systems are available these are incorporated in Agri-footprint,. To ensure the best time related representativeness, data will be updated regularly.

No specific choice of allocation approach is made, rather multiple options are presented to users. Two physical keys (dry matter and energy) and one economic allocation key are presented in Agri-footprint:

- Physical allocation: mass allocation
  For the crops and the processing of the crops, mass allocation is based on the mass of the dry matter of the products. For the animal products, mass allocation is based on the mass as traded.
- Physical allocation: gross energy allocation
  Water has a gross energy of 0 MJ/kg. The gross energy for protein, fat and carbohydrates are respectively: 23.6, 39.3 and 17.4 MJ/kg which are based on USDA (1973). Nutritional properties for gross energy calculations of products are based on a nutritional feed material list (Centraal veevoederbureau, 2010).
- Economic allocation
  For the crops and the processing of the crops the economic value of the products is based on (Vellinga et al., 2013).This allows users to choose an allocation that is most relevant for the context of their study. To ensure consistency in methodology and LCI modeling approach, Agri-footprint is reviewed by an independent external party.

System expansion is not applied in Agri-footprint because of the practical implications for the structure of the database. System expansion can be applied by the user by modifying processes. The allocation percentages of the unit processes can be set to 100% and 0% and the system can be expanded. In some specific situations avoidance of production is applied when the avoided product can be unambiguously determined such as electricity produced from a CHP delivered to a national grid.

2.2. Practical implementation – collection of data

The starting point of the inventory data development is information gathered in previous studies, particularly information gathered in the Feedprint project (Vellinga et al., 2013). The crop cultivation and processing inventories in that project contained only data to support carbon footprinting calculations. These inventories were further extended to include additional environmental flows to cover more impact categories. The additional flows were determined using a number of non-LCA models and data sources. Other environmental flows were added during the course of the development of Agri-footprint, to support assessment on other environmental impact categories. The LCIs have thus been extended to allow impact assessment on all impact categories of ReCiPe (Goedkoop et al., 2009) and ILCD (JRC-IES, 2011) impact assessment methods. Other impact assessment methods may also be supported, but naming of substances has aligned specifically to these two methods.

2.2.1. Crop cultivation

To expand the data from covering just greenhouse gas (GHG) related emissions, to coverage of a broader environmental scope, a number of models were developed, targeting specific environmental themes, relevant for all crop cultivation systems. These models focused on the themes land use change, fertilizer application, water use and heavy metal emissions. Other environmental themes were investigated using conventional literature or data survey, e.g. pesticide application rates (from a large volume of literature sources). How these models were used to create crop inventories is summarized below.

Fossil CO_2 emissions resulting from direct land use change are estimated using the "Direct Land Use Change Assessment Tool (Version 2014.1 - 21 January 2014)" that was developed alongside the PAS 2050-1 (BSI, 2012) and has been reviewed by the World Resource Institute (WRI) and WBCSD, and has, as a result, earned the ‘built on GHG Protocol’ mark. This tool provides a predefined way of calculating greenhouse gas (GHG) emissions from land use change based on FAO statistics and IPCC calculation rules, following the PAS 2050-1 methodology. GHG emissions arise when land is transformed from one use to another. This tool can be used to calculate these emissions for a specific country-crop combination and attribute them to the cultivated crops.

The tool provides means of estimating the GHG emissions from land use change based on an average land use change in the specified country. For Agri-footprint, the option “calculation of an estimate of the GHG emissions from land use change for a crop grown in a given country if previous land use is not known” was used. This estimate is based on a number of reference scenarios for previous land use, combined with data from relative crop land expansions based on FAOStat data (FAO, 2012). These FAO statistics then provide an estimate of the share of the current cropland (for a given crop) which is the result of land use change from forest and/or grassland to cropland. This share is calculated based on an amortization period of 20 years, as described in the PAS 2050-1. This results in three scenarios of land transformation (m2/ha*year): forest to cropland, grassland to cropland, and transformation between perennial and annual cropland, depending on the crop under study. The resulting GHG emissions are then the weighted average of the carbon stock changes for each of these scenarios. Further details and documentation can be found in the calculation tool itself (Blonk Consultants, 2014). The calculated CO_2 emissions from land use change (LUC) have been added in the database, the substance flow name is “Carbon dioxide, land transformation”. Note that land use change is also reported in m^2. By including the land use change in m^2, the impact categories ‘Land use: Soil Organic Matter (SOM)’ in the ILCD method and ‘natural land transformation’ in the ReCiPe method are supported.

Water use is based on spatially explicit water use methodology developed by Mekonnen & Hoekstra (2010). Water is used for irrigation of crops as well as during processing. The amount of (artificial) irrigation water is based on the ‘blue water footprint’ assessment (Mekonnen & Hoekstra, 2010b). The estimation of irrigation water is based on the CROPWAT approach (Allen, Pereira, Raes, Smith, & Ab, 1998). The blue water footprint refers to the volume of surface and groundwater consumed as a result of the production of a good. The model takes into account grid-based dynamic water balances, daily soil water balances, crop water requirements, actual water use and actual yields. The water footprint of crops have been published per country in m^3/tonne of product (Mekonnen & Hoekstra, 2010b). Combined with FAO yields (2007-2011) the blue water footprint is calculated in m^3/ha.

For the fertilizer application rates (in terms of kg NPK) the values of the Feedprint study are used. The majority of these fertilizer application rates are derived from data supplied by Pallière (2011) for crops in Europe, and data from Fertistat (FAO, 2011) for crops outside of Europe. Data from Pallière was preferred, because it was more recent. To match these total N, P and K application rates, to specific fertilizer types (e.g. Urea, NPK compounds, super triple phosphate etc.), a model was developed using data on country specific fertilizer consumption rates from IFA (IFA, 2012). In the analysis, it is assumed that the relative consumption rates of fertilizers are the same for all crops within a certain country. Some fertilizers supply multiple nutrient types (for example ammonium phosphate application supplies both N and P to agricultural soil). In IFA statistics, the share of ammonium phosphate is given as part of total N and also as part of total P supplied in a region. To avoid double counting, this dual function has to be taken into account.

Therefore the following calculation approach was taken: the fertilizers supplying K were considered in isolation. The relative shares of the K supplying fertilizers was calculated for a crop (e.g. if a crop A in Belgium requires 10 kg K/ha, 35% is supplied from NPK, 52% from Potassium Chloride and 11% from Potassium Sulfate). NPK however, also supplies a certain amount of N and P. The amounts of N and P supplied are subtracted from the total N and P requirements. Next, the share of P supplying fertilizers was calculated (however, the share of NPK in the P mix is not included as it is already accounted for during the calculation of K requirements). As in the P mix there are also fertilizers that contribute to the N supplied, this is also subtracted from the N requirements. For the remaining N required, the purely N supplying fertilizers are used (as NPK and ammonium phosphate are already considered during P and K calculations).

2.2.2. Processing of crops and animal products

Also, the starting point for crop processing was data gathered in the Feedprint project. Here the inventories have been extended to include non-greenhouse gas emissions, to cover more environmental impact categories. For example, additional data was gathered for water use in processing.

Unfortunately, the quality and availability of processing data is far from perfect. In some cases, the data are of quite good quality, meaning that they are representative for the region in scope, recently collected and accurate. In many other cases the data are of lower quality, and improvements are necessary.

Luckily some recent efforts by industry organizations are reducing this data quality issue. For example FEDIOL has recently published data on oil crushing and refining (Schneider & Finkbeiner, 2013). In addition, the Dutch meat processing industry has recently published data that was used to create inventories for slaughterhouse processes (for chicken beef and pigs) (COV & VNV, 2012). Also, some high quality company specific primary data on sugar processing from Suiker Unie has been collected and implemented in the database. As this producer of sugar and sugar products is the only major sugar producer in the Netherlands, the data is representative for Dutch sugar products. All this recent high quality data is included in Agri-footprint. Still collection of high quality processing data is an ongoing effort, and will be a main topic in future updates.

2.3. Data quality and review

Agri-footprint is externally reviewed by the Centre for Design and Society, RMIT University, Melbourne, Australia. This review is not a full ISO 14044 review. Rather, the external reviewers check the consistency and transparency of the methodology applied and completeness and transparency of data documentation.

PRé Consultants, the developers of SimaPro, did a thorough review to check technical database aspects, but also the transparency of the data (for example documentation and annotation of the Agri-footprint data within the software environment, and alignment of naming and structure to other databases already present in SimaPro).

3. Results

The first release of the database covers ~30 crops, ~100 (intermediate) products from processing, ~80 feed compounds, ~35 food products, 4 animal production systems and ~85 background processes (transport, auxiliary inputs for processing, fertilizers), with different versions for a number of countries/regions in the world and three pre-defined allocation systems. The total number of products in the database is ~3500. A full list of processes included in the database can be found on the Agri-footprint website (www.agri-footprint.com).

Also, two company specific LCIs are part of database, one for production of Nutramon (CAN) fertilizer from OCI, and sugar and sugar co-products produced by SuikerUnie.

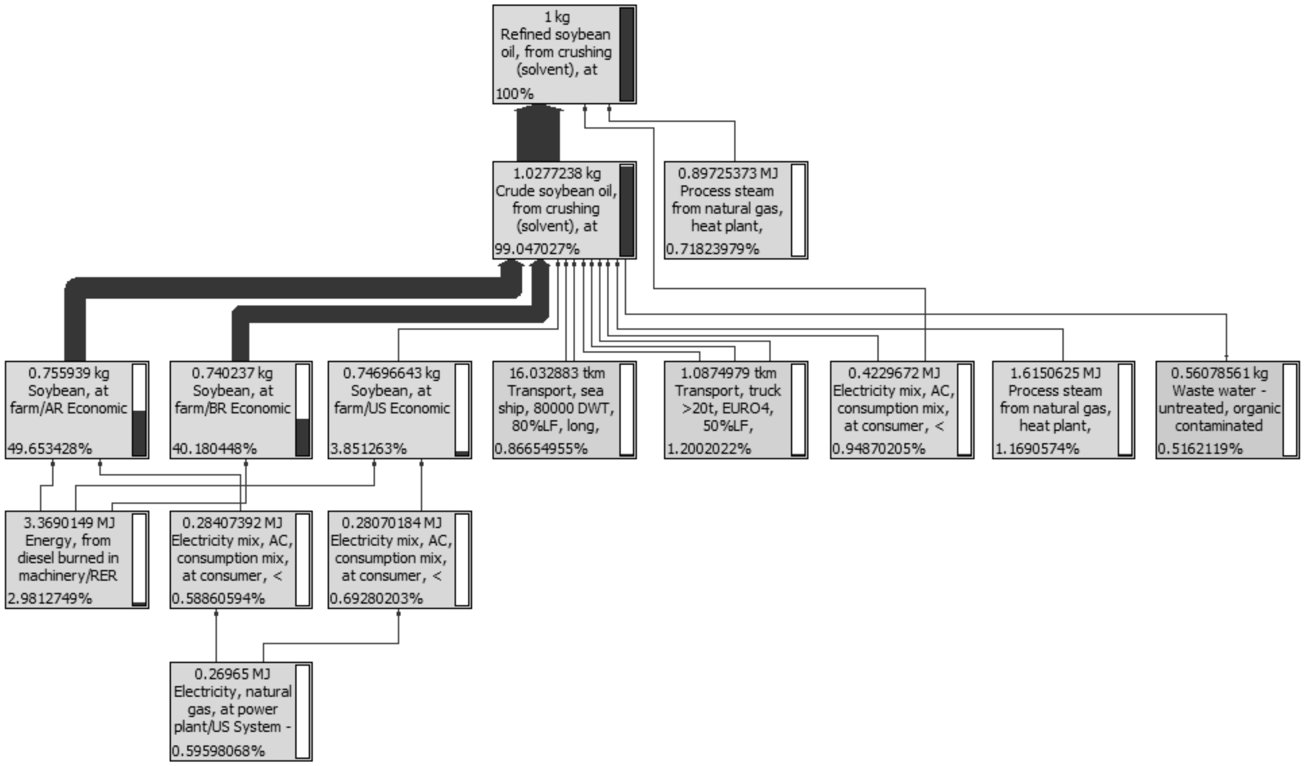

Figure 3. Example of a network diagram (in this case for refined soybean oil in the Netherlands) in Agri-footprint.

4. Discussion and conclusions

There are also a number of limitations that should be taken into account when using Agri-footprint. Internationally accepted LCA methodology for some agriculture specific environmental issues are still under development. For example, the methodology for assessment of loss of biodiversity due to land use or the estimation of direct and indirect land use change will probably evolve further. Due to limited methodology and data availability, elementary flows related to the environmental impact due to soil erosion and soil degradation are not included in Agri-footprint. The reliance on statistical data for crop yields, (artificial and organic) fertilizer application rates may not always appropriately reflect variability within regions and countries. Data availability of crop processing is also an issue that requires attention, as the use of older than desirable data sources and use of data from regions other than the region of interest is currently unavoidable.

To overcome these limitations, the database will be expanded and updated continuously. The aim is to yearly update data on crop yields, fertilizer use etc. Also the expansion to other crops and countries is important so that global production of some commodities can be better represented. Most importantly, the developers will continue to work with industry organizations to gather or gain access to new primary data for processing. Furthermore, additional non-LCA modeling approaches will be used to improve data quality on logistics and pesticide fates.

The database will be useful for LCA practitioners in the field of agricultural LCAs by providing relevant, transparent and consistent background data. As it provides an additional resource for LCA practitioners to use next to other currently available datasets, it will save time and facilitate the development of agriculture related LCAs. Agri-footprint will function as a platform for companies who wish to share their data. To ensure that the database will remain relevant in the future, there will be ongoing efforts to improve, expand and update the data.

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Environmental impacts of German food consumption and food losses

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ABSTRACT

Objective was to assess the environmental burden of food consumption and food losses in Germany with the aim to define measures to reduce environmentally relevant food losses. The assessment is based on a Screening Life Cycle Assessment and carried out from a consumer's perspective. Data have been taken from the German income and consumption sample and German production and trade statistics. In order to model the food baskets and its product chains some simplifications had to be done. The analysis shows that German food consumption emits 2.7 tons of greenhouse gases per person each year. 14 cubic meters of blue water are used for agricultural food production per person, and 2673 square meters of agricultural land are occupied each year per German for food consumption. In particular animal products cause high environmental burdens. Losses along the product chains have a share between 13 and 20 percent in environmental impacts.

Keywords: Screening Life Cycle Assessment, food consumption and losses, environmental impacts of German food basket, national origin of water and land use

1. Introduction

In recent years food waste is more and more of public interest. 2011 the documentary "Taste the Waste" came to German cinemas and its alarming message ("half of the food is spoiled") caused disgust about our way to deal with food in the public. In 2011 also the Food and Agricultural Organization (FAO) published a study on global food waste with the result that about one third of the food produced at global level is spoiled. This corresponds to 1.3 billion tons per year (Gustavsson et al. 2011). At the same time according estimations of the FAO 925 million people were starving.[1] Also in Germany a study on food waste was carried out on behalf of the Federal German Agricultural Ministry (BMELV). This study came to the result, that each year each German wastes 82kg food (Kranert et al. 2012).

In this context this study was carried out as part of a project aiming to reduce food waste in Germany on behalf of the Federal German Environmental Agency (Jepsen&Eberle 2014).

Aim of this part presented here was to assess the environmental burden of food consumption and food losses in Germany along the whole life cycle.

2. Methods

The assessment of German environmental impacts due to food consumption and food losses is based on a screening Life Cycle Assessment. Reference year is 2010.

The calculation is carried out from a consumer's perspective, meaning that the study starts from the consumer's food consumption (in-house and out-of-home) and analyzes every step of the various product chains. Drinks and candies haven't been considered. At the starting point the analysis differentiates between private consumers' food basket (in-house consumption) and the food basket of large scale consumers such as restaurants, and canteens (out-of-home consumption). Starting from in-house and out-of-home consumption the product chains are analyzed downstream as far as agricultural production. This includes storing and cooking in households respectively in canteens and restaurants, the shopping trip respectively food transportation to large scale consumers, retail and wholesale, food processing, agricultural production and all kind of transports along the products' life cycles. In order to provide information on the relevance and concrete impacts of food waste, environmental burdens are allocated to consumed and spoiled food on every step of the product chain. But for meat where food

[1] http://www.fao.org/mdg/goalone/en/; Status: 8. August 2012

losses due to slaughter by-products have been considered separately, all other food losses have been considered in the study without distinguishing between avoidable and unavoidable losses. [2]

2.1. System boundaries and modelling

The model was divided in four life cycle phases: agricultural production, food processing, retailing, and consumption.

System boundaries were set as follows:

For agricultural production energy, land, pesticide, fertilizers and water use for irrigation for production of plant products for direct human consumption but also for livestock feed have been taken into account. Land use and water consumption have only been taken into account in this life cycle stage. For livestock's breeding feed consumption, transports of feed and energy use were considered as well as direct emissions. Furthermore the necessary transports to food processing have been included within this life cycle phase.

For food processing energy use and direct emissions for all kinds of food processing like slaughtering, milling, baking, and processing of dairy products have been considered as have the transports to retail.

For retailing (wholesalers and/or retailers) energy use and freezing agent losses have been considered. Regarding in-house consumption also transports between wholesalers and retailers have been taken into account. Regarding out-of-home consumption transports from the wholesaler to the place of out-of-home consumption have been included.

For in-house consumption energy use for the shopping trip, the storing of purchased food, and cooking have been respected. For out-of-home consumption energy use for preparing of meals, food storing and air conditioning of restaurants have been taken into account; trips to the restaurant have been excluded. Customer transport to the place of out-of-home consumption has not been considered.

However, the production of seeds, water use outside agricultural production, all kind of packaging, and waste treatment haven't been considered within this study.

Modelling and the calculation itself were done with the software Umberto NXT LCA.

2.2. Data

Data for the composition of the two food baskets analyzed have been taken from statistical data such as the German income and consumption sample (Statistisches Bundesamt 2011) and German production and trade statistics (BMELV 2013). Also data from previous projects assessing the environmental impacts of German food consumption (Eberle et al. 2006, Wiegmann et al. 2005) have been used for this purpose.

Data for food losses have been used from two German studies carried out recently (Kranert et al. 2012, Peter et al. 2013) and from a study on behalf of the Food and Agricultural Organization (FAO) (Gustavsson et al. 2011).

Figure 1 and 2 show the material flows used as basis for the calculation of the environmental impacts of German food consumption.

Furthermore to calculate impacts it had to be estimated which share in the products was consumed cooked and which without cooking.

Most generic environmental data needed like electricity grids, transports, pesticides, and fertilizers have been taken from ecoinvent 3.01 database. Input and output data for agricultural production (including methane and nitrous oxide emissions from soils and animal production), food processing and retail have been taken from GEMIS 4.81 database. An exception has been made for water consumption data which are not included in GEMIS. These have been used from a study carried out by Mekkonen and Hoekstra (2010). In the study only data for the so called 'blue water' have been used.

To have the possibility to analyze where water consumption and land use is highest due to German food consumption and food losses, water consumption and land use in agricultural production were correlated with their national origins.

[2] This was done due to the reason that the characteristics 'avoidable' or 'unavoidable' are closely correlated with a value system that can change in the course of time.

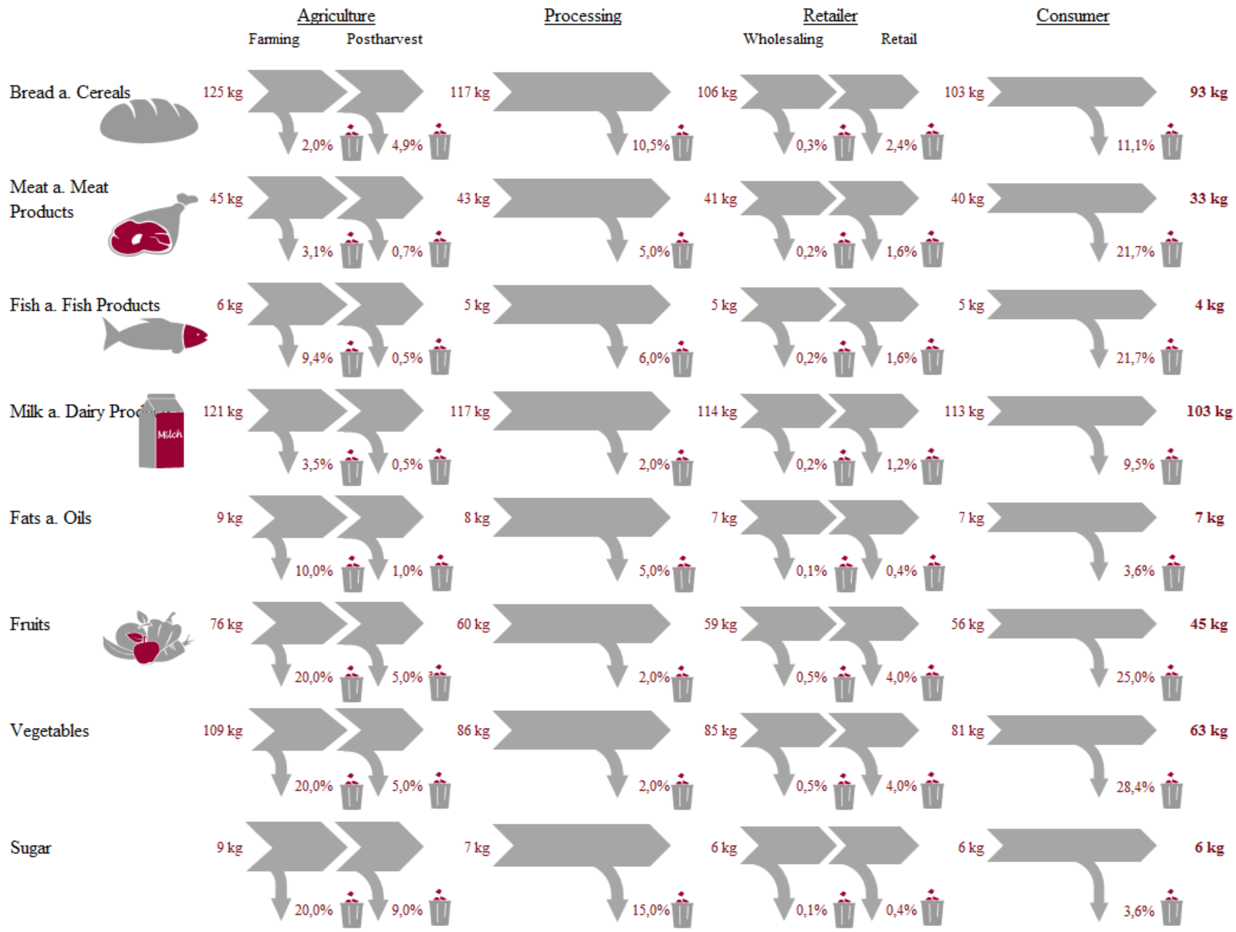

Figure 1. Material flows of in-house food consumption and food losses per person and year (number format: 0,2 = 0.2). Data are given in 'consumption' weight (e.g. boneless meat and w/o slaughter by-products).

Figure 2. Material flows of out-of-home food consumption and food losses per person and year (number format: 0,2 = 0.2). Data are given in 'consumption' weight (e.g. boneless meat and w/o slaughter-by products).

2.3. Simplifications and assumptions

In order to model the food baskets and its product chains some simplifications had to be done. Main reason for that was the lack of statistic and/or consistent environmental data, but also for modeling reasons (reduction of complexity of food production and distribution chains). The following simplifications have been made:

All food imports are modelled on agricultural level, thus also all food processing takes place in Germany. This simplification was done because statistical data don't show at which stage of the product's life cycle it is imported and also input/output data for processing in all the countries needed are not available.

Furthermore it was assumed that production of fodder components is done like in Germany with the same import countries and import shares for each fodder component. This simplification has to be done due to two reasons: one is the actual restrictions of the software program (which will be hopefully solved in next time); the other that import data for fodder components could not be looked into within this project. The composition of the livestock feed was modelled with respect to country specific data of the country where livestock breeding takes place.

Regarding poultry the assumption was made, that all meat is produced from broilers, the share of laying hens meat was not considered.

In addition, it was assumed that all food imports from overseas are carried out only by ship. This simplification was done with respect to the very low relevance of air freight transports of food to Germany which is about 0.12% of all food imports (Keller 2010).

Moreover, organic production systems are not included also due to their low relevance – only 6% of the agricultural land in Germany is cultivated regarding standards for organic production (BLE 2012) – and due to the fact that not all data needed have been available for organic production.

Besides, it was assumed that households buy all their food at retailers, purchases direct at the farm or at local markets haven't been considered. For out-of-home consumption it was assumed that all food is delivered by wholesalers.

2.4. Allocations

Most agricultural production systems have more than one output. Milk cows, for example are kept for milk as main product and meat as co-product. In LCAs environmental burdens need to be allocated to the products by different allocation methods (mass, economic, or commodity specific allocations). In order to reduce the complexity of the model and due to some data lacks allocations have been made only regarding food losses, where a mass allocation was applied at all life cycle stages. In case of meat a physical allocation was done to allocate burdens to meat and slaughter by-products. An economic allocation was renounced because prices in particular regarding by-products vary enormously with respect to time and geographical origin. Thus, regarding milk cow keeping an economic allocation was made with the result that 80% of impacts have been allocated to milk.[3] Also with respect to dairy production allocations have been made. Here an allocation with respect to milk solids was chosen, which is regarded as 'fairest' allocation method for dairy products (Lundie et al. 2007). Regarding the production of soy and rapeseed shred material flows have been allocated to oil and shred regarding their heating value equivalents, which was the approach chosen in the database used. In all other cases of agricultural production 100% product allocation was chosen. This approach leads to a slight overestimation of environmental burdens in agriculture.

In case of combined power generation burdens were allocated in relation to energy yield.

2.5. Impact assessment

The impact assessment methodology used is ReCiPe Midpoint (Goedkoop et al. 2008). The following environmental impact categories have been assessed: climate change, fossil depletion, freshwater eutrophication, marine eutrophication, metal depletion, ozone depletion, particulate matter formation, photochemical oxidant formation, and terrestrial acidification. Furthermore also the use of agricultural land and agricultural water use for food production have been analyzed. Toxicity indicators have not been assessed mainly because the input data for pesticide use available for the different foods have been very unspecific.

3. Results

The analysis shows that German food consumption emits 2.7 tons of greenhouse gases per person each year. 14 cubic meters of blue water are used for agricultural food production per person, and 2673 square meters of agricultural land are occupied each year for each German for food consumption. Table 1 shows total results for each indicator according to life cycle phases.

The results show that agricultural production and consumption are responsible for the main impacts of German food consumption and food losses. For all indicators analyzed these two life cycle phases cause more than 87 percent of the environmental burden. In contrast food processing and retailing have for all indicators and inventory parameters less environmental impact.

GWP-100, fossil depletion, freshwater and marine eutrophication, metal depletion and terrestrial acidification are mainly caused by energy use along the products' life cycles and in particular GWP-100 also by emissions directly from agricultural production. Particulate matter formation and photochemical oxidant formation originate mainly from transport emissions.

The biggest part of the impacts is caused by in-house consumption (61-80% for each indicator) which has also the highest share in amounts. Food losses due to in-house food consumption have a share in the total environmental burden between 8 and 14 percent, out-of-home food consumption 6 to 19 percent, and food losses due

[3] The remaining 20% have been allocated to mother cow meat, but haven't been considered further in the model.

to out-of-home consumption between 2 to 9 percent. Total losses have a share of 15 to 21 percent. Figure 3 shows the shares in in-house and out-of-home food consumption and food losses regarding environmental burdens of the different indicators and inventory parameters.

Regarding the consumed respectively wasted products results show that animal products like meat and dairy products cause most of the environmental burden of food consumption and food losses, although the share of plant products is higher regarding amounts of consumption or waste. This is the case for all analyzed impact indicators. Only regarding agricultural water use, plant products consume more water in total and per kilogram product (Table 2, Table 4).

Table 1. LCA results for life cycle phases and in total per person and year

Impact categories	Unit	Agriculture	Processing	Retailing	Consumption	Total
GWP-100a	kg CO2e	1.56E+03	1.19E+02	5.81E+01	1.02E+03	2.75E+03
Fossil depletion	kg oil-equiv.	2.69E+02	2.56E+01	1.51E+01	4.50E+02	7.59E+02
Freshwater eutrophication	kg P-equiv.	1.64E-01	8.16E-02	5.45E-02	7.33E-01	1.03E+00
Marine eutrophication	kg N-equiv.	9.25E-01	1.57E-01	1.44E-02	2.09E-01	1.31E+00
Metal depletion	kg Fe-equiv.	2.23E+01	1.13E+00	1.00E+00	4.51E+01	6.96E+01
Ozone depletion	kg CFC-11-equiv.	5.90E-05	5.72E-06	3.20E-06	1.25E-04	1.93E-04
Particulate matter formation	kg PM-10-equiv.	2.60E+00	6.81E-02	3.13E-02	9.79E-01	3.68E+00
Photochemical oxidant formation	kg NMVOC	9.66E+00	4.19E+00	3.79E+00	4.51E+01	6.27E+01
Terrestrial acidification	kg SO2-equiv.	1.48E+01	2.00E-01	8.09E-02	2.58E+00	1.76E+01
Agricultural land use	m²*a	2.67E+03	0.00E+00	0.00E+00	0.00E+00	2.67E+03
Agricultural water use	L	1.40E+04	0.00E+00	0.00E+00	0.00E+00	1.40E+04

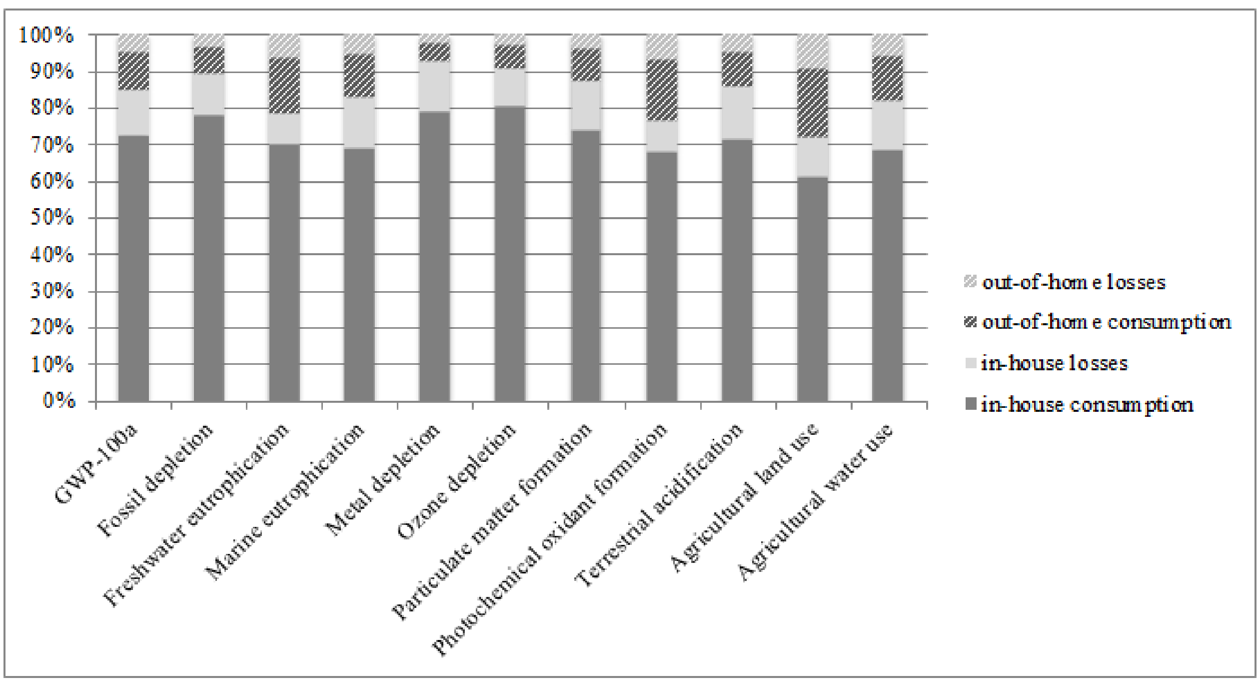

Figure 3. Shares of in-house and out-of-home food consumption and food losses regarding environmental burdens caused by German food consumption

Results per kg product (Table 2) show that animal products in the German food basket have a higher impact for all analyzed impact categories and parameters than have plant products in the German food basket. The only exception is water use. In particular in the case of agricultural land use for food production this is obvious: for the production of animal products eight times more land is needed per kilogram food than for plant products.Also with respect to the indicator terrestrial acidification differences are significant: the impact per kg consumed animal based food is nine times higher than that of products with a plant based origin. Also for marine eutrophication (six times higher), particulate matter formation (five times higher), and global warming (four times higher) the differences are similar. For all other indicators the impact of animal products is between 1.7 and 4.7 times higher than that of plant products. Only for agricultural water use (irrigation water) for food production the water use of animal products is lower as for plant products.

Results show that in total most water for German food consumption and losses is used in Germany (23%) followed by Spain (18%) and Pakistan (17%). Results for animal products show that also Germany is responsible for most of the water use (77%) followed by France (11%) and Argentina (8%). In contrast regarding plant based food most water is used in Spain (23%) followed by Pakistan (23%) and the US (9%).

Regarding land use most agricultural land is used in Germany (73%). This is also the case for animal (70%) and plant based food (75%). Germany is followed by Argentina and Brazil (both 9%). This is the same for animal-based products but not for plant-based products where the next highest shares have the Netherlands and the Czech Republic with 5 percent.

Table 3 shows the additional environmental impacts which are caused per kilogram consumed food due to food losses. These are much higher for out-of-home consumption than for in-house consumption[4]. This is mainly due to the fact that losses at out-of-home consumption are much higher than for in-house consumption but also for the differences in the composition of the consumed food. Thus, the high value for per kilogram consumed food for water use is mainly due to the fact that at out-of-home consumption regarding our data much more rice is consumed and spoiled than at in-house consumption. The value for land use is so much higher because of the higher share of waste but also due to the higher consumption and in consequence losses of meat.

Table 2. Average impact for German food consumption and losses per kg product

Impact categories		Animal products	Plant products
GWP-100a	kg CO2e	9.21	2.55
Fossil depletion	kg oil-equiv.	2.10	1.00
Freshwater eutrophication	kg P-equiv.	2.78	1.41
Marine eutrophication	kg N-equiv.	4.92	0.85
Metal depletion	kg Fe-equiv.	1.74	1.04
Ozone depletion	kg CFC-11-equiv.	0.53	0.26
Particulate matter formation	kg PM-10-equiv.	1.32	0.28
Photochemical oxidant formation	kg NMVOC	1.69	0.86
Terrestrial acidification	kg SO2-equiv.	7.15	0.80
Agricultural land use	m²*a	10.66	1.34
Agricultural water use	l	1.89	3.17

[4] Blumenthal&Göbel (2014) found out that in German communal feeding food losses add to 8 to 30 percent of food consumption in this sector. According to our data the share in food losses is 33.5 percent in out of home consumption (see fig. 2).

Table 3. Additional environmental impacts due to food losses per kg consumed food

Impact categories		in-house consumption	out-of home consumption	total consumption
GWP-100a	kg CO2e	0.9	2.8	1.1
Fossil depletion	kg oil-equiv.	0.2	0.6	0.3
Freshwater eutrophication	kg P-equiv.	2.3E-04	1.4E-03	3.6E-04
Marine eutrophication	kg N-equiv.	4.8E-04	1.5E-03	6.0E-04
Metal depletion	kg Fe-equiv.	2.5E-02	3.6E-02	2.6E-02
Ozone depletion	kg CFC-11-equiv.	5.2E-08	1.2E-07	6.0E-08
Particulate matter formation	kg PM-10-equiv.	1.3E-03	3.2E-03	1.5E-03
Photochemical oxidant formation	kg NMVOC	1.3E-02	9.4E-02	2.2E-02
Terrestrial acidification	kg SO2-equiv.	6.6E-03	1.8E-02	7.8E-03
Agricultural land use	m²*a	0.8	5.3	1.3
Agricultural water use	L	4.8	18.1	6.3

Table 4. National origin of water and land used for agricultural food production per person and year[5]

Country	Unit	Animal products	Plant products	Total	Unit	Animal products	Plant products	Total
Argentina	l	282	0	282	m²*a	216	0	216
Austria	l	0	9	9	m²*a	0	1	1
Brazil	l	44	0	44	m²*a	216	0	216
Colombia	l	0	20	20	m²*a	0	2	2
Croatia	l	0	344	344	m²*a	0	2	2
Czech Republic	l	2	7	10	m²*a	25	20	45
Germany	l	2556	712	3268	m²*a	1353	317	1669
Denmark	l	6	9	15	m²*a	3	2	5
Ecuador	l	0	503	503	m²*a	0	3	3
Egypt	l	0	63	63	m²*a	0	0	0
Spain	l	0	2521	2521	m²*a	0	8	8
France	l	381	16	397	m²*a	15	12	27
Hungary	l	1	0	1	m²*a	4	0	4
Israel	l	0	160	160	m²*a	0	0	0
India	l	0	300	300	m²*a	0	2	2
Italy	l	0	647	647	m²*a	0	6	6
Maroc	l	0	15	15	m²*a	0	0	0
Netherlands	l	52	23	75	m²*a	9	22	31
Pakistan	l	0	2437	2437	m²*a	0	3	3
Poland	l	8	3	10	m²*a	17	13	30
Swaziland	l	0	12	12	m²*a	0	0	0
Thailand	l	0	884	884	m²*a	0	5	5
Turkey	l	0	126	126	m²*a	0	0	0
United Kingdom	l	1	712	713	m²*a	2	2	4
United States	l	0	925	925	m²*a	0	3	3
Viet Nam	l	0	293	293	m²*a	0	4	4
TOTAL	l	3333	10740	14073	m²*a	1861	425	2286

[5] Interpreting the results it has to be considered that results for animal based products depend on the assumption that production of fodder components is done like in Germany with the same import countries and import shares for each fodder component. Thus, shares of Germany are somewhat overestimated with respect to water and land use (see sections 2 and 4).

4. Discussion

Results show a high relevance of food consumption and food losses regarding environmental impacts: e.g. food consumption and food losses cause about 23 percent of the German greenhouse gas emissions per person[6] and the water used for German food production is about one third of the German households' water use (Destatis 2013).

In general, results show a similar dimension as results from previous studies which have been carried out to estimate environmental impacts of German food consumption. However, there are also differences. One reason for that is, that in this study both the whole life cycle from agriculture to consumption (including energy consumption for shopping trip, food storage and cooking) and also food losses at all life cycle stages have been considered which was not the case in previous studies

Wiegmann et al. (2005) calculated greenhouse gas emissions which are one quarter lower. They used a similar methodology but a different database. The results of Meier (2014) for greenhouse gas emissions are 9 percent lower. Meier also used a different database.

There are two main reasons for the differences: one is that Wiegmann et al. (2005) didn't calculate all food losses along the value chain, because data haven't been available in sufficient detail at that time. Meier (2014) didn't calculate energy consumption at household level for the shopping trip, cooling and cooking. The other reason is that both studies used the GEMIS database for basis data as electricity grid, fertilizer and chemicals production, transport. In contrast, in this study the ecoinvent database was used for basis data, GEMIS data were only used for material flows. Compared to the ecoinvent database greenhouse gas emissions in the GEMIS database are lower in most cases.

Regarding water consumption Meier (2014) calculated much higher water consumption. According to his results German food consumption is responsible for 32.5 m³ of water use per person and year which is more than twice as much as the results of this study. The main reason for that is that in Meier's study (Meier 2014) nuts count for about one third of water consumption of German food consumption. In this study nuts haven't been an own category, they have been subsumed in the category 'other fruits' and thus specific water use of nuts has not been taken into account. This and also other differences regarding the composition of the calculated food baskets explain differences in agricultural water use.

In contrast, with respect to land use results of Meier (2014), Wiegmann et al. (2005) and Kastner et al. (2012) are about 10% lower. There are two main reasons for this difference. One is that different data regarding land use have been used in the studies; the other that food losses have not been taken into account in all studies. Yield data in this study have been taken from GEMIS 4.81[7], which uses yield data from the Common Agricultural Policy Regionalized Impact Analysis (CAPRI) modelling system[8] in most datasets. GEMIS groups countries to Central, North, West and South Europe, and Germany has been assigned to Central Europe. Probably this leads to lower yields than typical for Germany.

Furthermore results of this study have to be discussed against the allocation methods used. For this purpose a sensitivity analysis has been carried out to analyze the influence of the allocation method used for milk cow keeping and processing of dairy products. In the sensitivity analysis both allocations have been changed to 100% allocation. Sensitivity analysis results show that this allocation influences results. With respect to all analyzed impact categories and inventory parameters results are 3 to 19% higher with 100% allocation. In particular regarding greenhouse gas emissions (8%), particulate matter formation (95%), terrestrial acidification (19%) and agricultural land use (18%) an effect of the chosen allocation can be shown (Figure 4).

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[6] http://www.umweltbundesamt.de/themen/klima-energie/klimaschutz-energiepolitik-in-deutschland/treibhausgas-emissionen/europaeischer-vergleich-der-treibhausgas-emissionen; Status: 8 May 2014
[7] http://www.gemis.de
[8] http://www.capri-model.org/

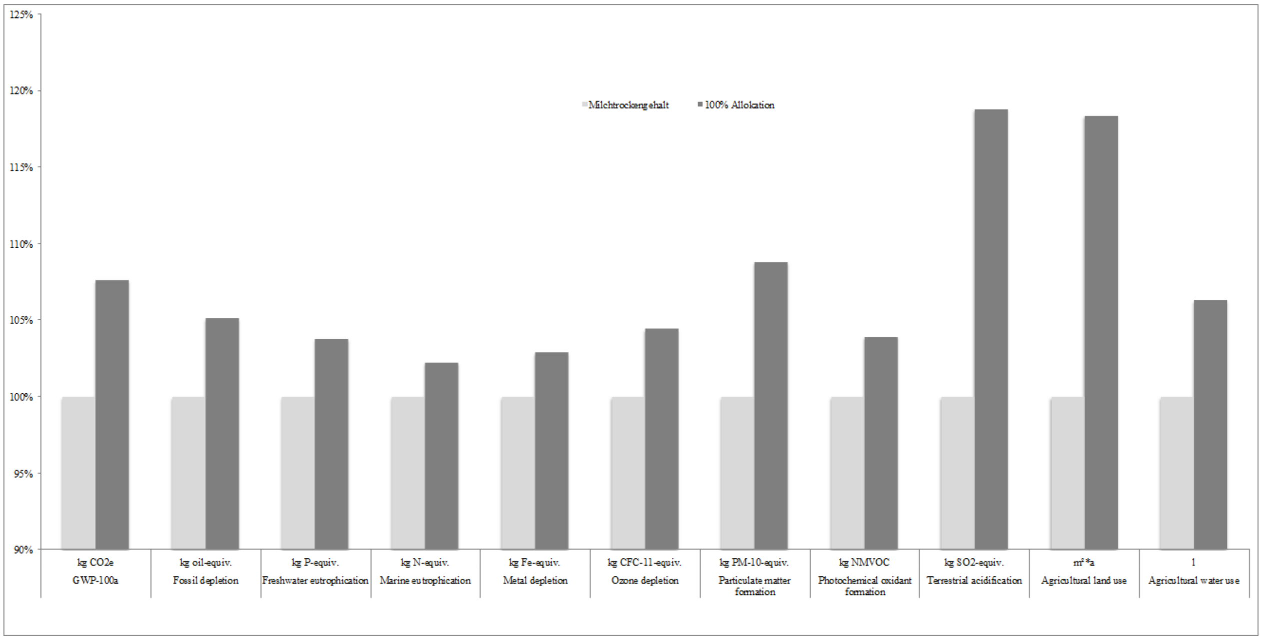

Figure 4. Differences in results with respect to allocation methods

Moreover, results have to be discussed against the assumptions and simplifications made. Thus, with respect to the chosen methodology it has to be considered that results for animal based products depend on the assumption that production of fodder components is done like in Germany with the same import countries and import shares for each fodder component. Thus, shares of Germany are somewhat overestimated with respect to water and land use.

Regarding the provenience of water consumption it has to be taken into account that in the case of rice no statistical data for the origin of consumed rice could be found. German trade statistics show only the import countries of processed rice (e.g. peeled rice). Thus, it was assumed that the world's largest rice exporters export rice to Germany at the same proportion as their share in the global rice market. Therefore, more precise trade statistics could change the results in the case of Pakistan.

5. Conclusion

The study shows the high relevance of food production regarding environmental impacts. In particular animal products are responsible for high environmental burdens in the German food basket. Losses (animal and plant based) along the product chains have a share between 13 and 20 percent in environmental impacts. With respect to reduce environmentally relevant food losses, measures should focus in particular to reduce food waste of animal origin like dairy products and meat. The most relevant points for reduction measures are agricultural production and consumption in households and out-of-home. In particular out-of-home consumption has a high share of spoiled food in relation to the total food used. Out-of-home consumption therefore also provides a good starting point for measures.

Nevertheless, better statistical and also environmental data are still needed to improve such kind of studies.

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Progress Report: Methodology of Chilean Food & Agriculture LCI Database

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ABSTRACT

The lack of harmonized and regionally appropriate life cycle inventory databases is one of the main challenges limiting the use of life cycle analysis for decision making on food systems. The food and beverage sector in Chile is important for the country and is strongly focused on exports, where requirements for sustainable practices and value chain transparency are increasing, but there is a lack of technical capacity within companies to analyze and interpret sustainability assessment. In this context, Fundación Chile and Universidad de la Frontera are developing the EcoBase Food project, an environmental information management system for life cycle analysis, through a technological platform, to improve the sustainability and competitiveness of the food and viticulture exports industry. The following document presents a summary of the data collection methodology to create the life cycle inventories for the project, which corresponds to an adaptation from international best practices and is aligned to international initiatives.

Keywords: life cycle analysis, life cycle inventories, Chilean food and viticulture sector, LCI methodology, LCA tools

1. Introduction

The lack of harmonized and regionally appropriate life cycle inventory (LCI) databases is one of the main challenges limiting the use of life cycle analysis (LCA) for decision making on food systems (Heller, Keoleian and Willett, 2013). Several LCA and environmental information initiatives have been developed in recent years, but there is little coordination among them. Some of these initiatives are:

Table 1. Initiatives and sources of information

Type	Initiative	Information
Users of data	The Sustainability Consortium	http://www.sustainabilityconsortium.org/sp/
	Product Environmental Footprint	http://ec.europa.eu/environment/eussd/smgp/product_footprint.htm
	Nordic Swan	http://www.nordic-ecolabel.org/
	Grenelle Law	http://www.developpement-durable.gouv.fr/IMG/pdf/Grenelle_Loi-2_GB_.pdf
Database development initiatives with different methodologies	World Food Database	http://www.fcrn.org.uk/research-library/food-and-its-life-cycle/other-studies/world-food-lca-database
	International Life Cycle Data System (ILCD)	http://eplca.jrc.ec.europa.eu/?page_id=86
	Ecoinvent	http://www.ecoinvent.org/database/
	AusLCI	http://alcas.asn.au/AusLCI/
	AgriBALYSE	http://www2.ademe.fr/servlet/getDoc?id=38480&m=3&cid=96
	LCA Digital Commons	https://www.lcacommons.gov/nrel/search

The food and beverage sector in Chile is an important sector, strongly focused on exports. It is the second largest sector for the Chilean economy and ranks 15th worldwide (US Commercial Service, 2012). Requirements for sustainable practices and value chain transparency are increasing (ProChile, 2012), both from international retailers (Walmart, Tesco, among others) and from within the country as a member of the OECD (OECD, 2005). Along with this, there is a lack of technical capacity within companies in Chile to analyze and interpret sustainability assessment.

In this context, Fundación Chile and Universidad de La Frontera (UFRO) are developing the EcoBase Food project, which has secured funding from a public grant presented with the following title: "Development of an Environmental Information Management System for Life Cycle Analysis, through a Technological Platform, to improve the sustainability and competitiveness of the Food and Viticulture Exports Industry". The project relies on the support and interest of 9 different trade organizations from the food and wine sector, along with the Environment and Agriculture ministries and the Chilean Promotion Bureau (ProChile due to its Spanish initials).

The aim of the project is to increase the competitiveness of food exporters and producers, including small to medium enterprises (SMEs).It is expected that companies will be able to increase their competitiveness by:

- Providing transparent and rigorous product sustainability information (e.g. environmental footprints) to clients.
- Demonstrating positive and competitive sustainability performance.
- Identifying hotspots, evaluating solutions and prioritizing sustainability improvements based on this information.

To achieve this, the project will deliver an environmental information platform, which will provide:

- A methodological framework and guidelines for consistent data collection and impact assessment
- Excel based data collection tool.
- 16 national product category (Table 2) data baseline models, with options to model different production systems per category.
- Instructions for selecting indicators, best practices and improvement opportunities associated with major environmental impacts.

The following document presents a summary of the data collection methodology to create the LCIs for food and wine products in Chile. The methodology is an adaptation from international best practices and aligned to international tools, standards and recognition schemes; balancing the generality and specificity of these to suit the Chilean industry requirements the best. The main sources for the methodology are:

- The International Organization for Standardization’s Life Cycle Assessment method (International Organization for Standardization (ISO) 2006a; International Organization for Standardization (ISO) 2006b)
- Ecoinvent Quality Guidelines (Weidema et al., 2012), as an important centre of expertise and support for the project, as well as the potential to provide the Ecoinvent database with information from the project.
- Shonan Guidance Principles (Sonnemann and Vigon, 2011), as a general guide for the methodology structure (Context, Unit process, Aggregation, Documentation, among others).

Along with the above, the methodology has been developed taking into account the different opinions of the trade organizations and ministries involved in the project, and will pass through a critical review before final publishing.

The methodology could be applicable to any food or beverage product, but the project will focus on 16 different product groups (Table 2), which are categorized into: Fruits & Wine, Aquaculture and Meat & Dairy. Collectively, in 2012 the considered product categories represented 62% of Chile’s food and beverage exports (Chilean Central Bank, 2012).

Table 2. Product categories in the project

Fruits & Wine	**Aquaculture**	**Meat & Dairy**
• Fresh Apples	• Salmon	• Pork
• Dried Apples	• Mussels	• Chicken
• Table Grapes		• Powdered milk
• Apple Juice		• Gouda cheese
• Wine		
• Avocado		
• Fresh Plums		
• Fresh Blueberries		
• Canned Peaches		
• Frozen Raspberries		

This project is being developed in parallel with the construction sector, in pursuit of a national project that aims to have a coordinated platform of information for relevant industries of the Chilean economy. Having databases which are coordinated and have similar structures facilitates information exchange and allows for improved decision-making at a country level, optimizing and making resource use more efficient. With the objective of having these projects as part of a more harmonized international network, Fundación Chile has become a member of UNEP/SETAC´s Life Cycle Initiative, and is actively participating in international developments.

2. Methodology

The present methodology aims to enable the quantification and reporting of environmental impacts associated with the agricultural/livestock/aquaculture production and processing stages, linked to upstream and downstream data. The project will create “cradle to gate" inventories and offer tool support to model “cradle to retail gate” scenarios.

The document includes:

- A methodological framework providing consistent data collection and impact assessment guidelines;
- Defined product categories (Table 2), with their respective lists of processes, inputs and outputs to facilitate data collection.

2.1. Scope

The goal of the project is to increase the competitiveness of food producers and exporters, including SMEs. It also aims to obtain better understanding of the impacts of the Chilean food and wine supply chains and to introduce and enable the industry to use life cycle thinking.

Therefore, one of the target audiences is exporting companies from the food and wine sector. Another key envisioned target audience for the sustainability information is future retailers at key export markets (e.g. French retailers in coming years). Therefore the functional units and scope need to be appropriate for use by retailers and cradle to retailer gate, which is why the functional unit will be a unit of mass or volume rather than a nutritional unit for example. This is useful from the consumer’s point of view but not from producers and retailers standpoint.

To this end, the methodology aims to generate national LCI baseline models that could be later used as a base for more complex LCA studies. Also, it will try to generate unit process inventories as disaggregated as possible, protecting minimum requirements of confidentiality.

The functional unit will be 1 kg of packaged product, except for wine which will be 0.75 l of packaged wine, since this is the most purchased format. Additionally, the most representative varieties or types of products (e.g. Royal Gala and Granny Smith apple varieties) of the Chilean industry will be employed. These measures will allow having consistent, comparable and representative information.

2.2. System boundaries

The stages covered are: i) raw materials, ii) agriculture, iii) livestock & aquaculture, iv) processing, v) packaging and vi) distribution.

A list of inclusions and exclusion per life cycle stage is given in Table 3.

Table 3. Life cycle stage inclusions and exclusions

Life cycle stage	Inclusions	Exclusions
Raw materials (secondary data)	• Agrochemical production • Other chemical input • Fuel production • Transportation[a]	• Manufacture, transportation, and end-of-life disposal of buildings, transportation and packaging infrastructure, capital equipment and auxiliary machineries • For crop production, it is not necessary to include the production of seeds as the relative contribution to environmental impact is

		small. The exception to this is where seed production would be expected to be, or shown to, represent more than 1% of the impact to any impact category (for example where small organic farms may be studied).
Agriculture	• Water use • Land use • Soil carbon sequestration • Agrochemical and fertilizers use • Fuels combustion and electricity • Machinery • Waste management • Transportation[a]	• Manufacture, transportation, and end-of-life disposal of buildings, transportation and packaging infrastructure, capital equipment and auxiliary machineries • Consumables • Staff transport • Noise and vibration, radiation, odor and waste heat
Livestock & Aquaculture	• Feed • Fuels combustion and electricity • Chemicals • Fugitive emissions • Water use • Land use • Soil carbon sequestration • Agrochemical use • Waste management • Transportation[a]	• Manufacture, transportation, and end-of-life disposal of buildings, transportation and packaging infrastructure, capital equipment and auxiliary machineries • Consumables • Staff transport • Animal welfare • Noise and vibration, radiation, odor and waste heat
Processing	• Fuels combustion and electricity • Chemicals • Fugitive emissions • Water use • Waste management • Transportation[a]	• Manufacture, transportation, and end-of-life disposal of buildings, transportation and packaging infrastructure, capital equipment and auxiliary machineries • Consumables • Staff transport • Noise and vibration, radiation, odor and waste heat
Packaging	• Source of plastics, paper, glass, aluminium, metals, etc. • Recycled content • Primary, secondary & tertiary packaging • Transportation[a]	• Manufacture, transportation, and end-of-life disposal of buildings, transportation and packaging infrastructure, capital equipment and auxiliary machineries • Consumables • Staff transport • Noise and vibration, radiation, odor and waste heat
Distribution	• Storage; • Distribution to retail outlets; • Transportation to retail outlets; • Cold storage and preservation processes	• Manufacture, transportation, and end-of-life disposal of buildings, transportation and packaging infrastructure, capital equipment and auxiliary machineries • Noise and vibration, radiation, odor and waste heat

[a]Transportation shall include all modes (road, rail, air, and by ship) occurring in-between the life cycle stages.

2.3. Allocation procedure

In LCA, environmental impacts and benefits are calculated for different materials, products, assemblies and services. These products are produced by a large diversity of production systems that make up a modern industrial economy. Whenever a single process produces more than one product or service (e.g. beef being produced as a co-product in the production of milk), an approach is needed to determine how the environmental impacts of the single process should be assigned to each of the products or services. This is commonly referred to as the allocation problem.

For the EcoBase project, sufficient data shall be collected to enable flexibility with regards to allocation between co-products. This includes co-product price information, mass/volume, etc. Where allocation between co-products cannot be avoided or based on direct causality, economic allocation will be used, in order to allocate inputs and outputs between the products and functions in a way which reflect their value. This is in line with the Handbook on LCA, which advises economic allocation as baseline method for most situations (Guinée et al., 2002).

This approach also allows allocating impacts between recycled products and waste-derived fuels.

2.4. Cut-off rules

No strict quantitative cut-off rule is followed in the project. The database shall be as complete as the knowledge of the data providers allow. In principle, all known inputs and outputs are recorded as such. All input and outputs within the assessment scope shall be included as far as practical and reasonable. However, for the purpose of prioritizing efforts, the following threshold can be applied:

- Strong efforts to collect high quality data shall be held when inputs are anticipated to make a material contribution (more than 1% normalized impact in the priority impact categories) to the total life cycle inventory per functional unit.
- Otherwise, data can be collected from conservative estimates.
- Where a single source of energy and water use and emissions accounts for more than 50% of the likely total life cycle inventory per functional unit, the above threshold rule (inclusion of at least 95%) shall apply to the remaining life cycle inventory.

Further information on data sources is given in section 2.5 below.

2.5. Data collection guidelines and tool design

For this project, one of the goals is to obtain data that represents the various realities (production technologies, raw materials, etc.) of each product. It will work with the principle of collecting the best available information, prioritizing completeness of systems, and basing the data collection efforts according to the following hierarchy of sources of information (adapted from Howard & Sharpe, 2010):

1. Primary information: Information obtained directly from companies
2. Secondary information:
   - Studies and other national projects related to each category.
   - Information available from the Clean Production Agreements from the National Council of Clean Production
   - Information available in the Chilean national environmental impact assessment database (e-SEIA).
   - Documented estimates from national experts.
   - International studies or other projects related to each category which could be complemented with international experts' contributions.
   - LCA databases internationally recognized, with the necessary and feasible adjustments to adapt them to the Chilean reality (energy matrix, transport distances, etc.).

Based on feedback by the trade associations supporting the project, the data collection tools developed have been designed to increase usability by the data providers. For each of the products in the database, a process list, with its inputs and outputs listed has been generated to guide the data collection process. These process lists are included within the data collection tools to facilitate the navigation and filling of the spreadsheets. Also, when information is available from literature review, some values are pre-filled to assist the data providers in case they don't have certain fields of information, or if they feel that the pre-filled values represent their production process; validating them. The data collection tools will be sent to private companies, through their respective trade association, with the project teams assisting the filling of the spreadsheets through visits to the facilities or farms, online workshops or other ways of more informal interaction.

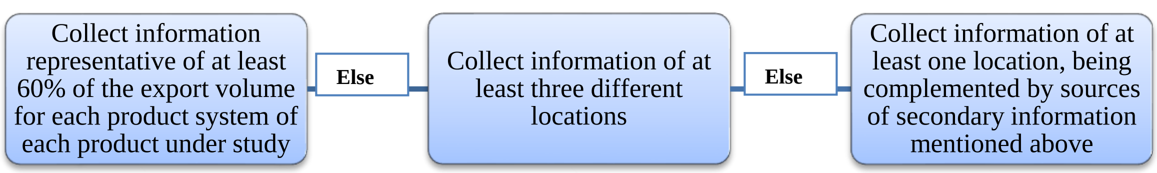
Data collection guidelines for each impact category have been developed and prioritized, based on literature and contribution analysis for several of the product categories. Data collection efforts will be focused according to the representativeness goals given in section 2.5.1 and the hierarchy of sources of information.

2.5.1. Representativeness

In order to maximize the usefulness of the LCI developed, a key aspect is representativeness (from a geographic, technological and productive point of view) of the most important product systems in term of market share exports.

Therefore, a hierarchy of objectives was defined and it is presented below

Figure 1: Hierarchy of objectives



For all cases, the export volume and relevance of the productive systems for national exports will be considered for selecting each location.

2.5.2. Averaging, aggregation and data confidentiality.

As one of the objectives of the project is to provide national baseline LCIs for the products under study, it is necessary to appropriately average and aggregate data from different companies.

In order to protect the confidentiality of individual company's data, the Project will allow for data to be published as aggregated from several sources into anonymous generic averages that are nonetheless representative of the Chilean industry. In general, inputs and outputs of several distinct unit processes are aggregated only if a) data is not available on a detailed unit process level, or b) unit process data is confidential.

The project will average data horizontally. This means that average inventory for each of the individual subprocesses in an inventory are weightily averaged and the sum of these then represents the inventory for the whole. This is preferred because it preserves data richness and it can be used to compare between similar processes from different industries. Also, it can even be used as a proxy process in another sector to fill a non-critical data gap. Maintaining data richness provides advantages such as the ability to see which parts of the processes contribute more to the impacts of their products. This can further help understand how to innovate to mitigate those impacts.

Two particular cases need special attention:

- Only one surveyed company: When only one company is surveyed in a product category, confidentiality will be ensured by combining their data with information from other national or international databases that are considered suitable.
- Various production locations per company: If a company has more than one productive location, LCI averages for each location will be weighted according to the percentage of production of the whole company. Special care will be held when different productive technologies exist inside one company.

The Project will estimate and document the extent to which data are representative of a market or a process, but will not exclude data that cannot achieve any particular threshold level of representativeness. The data used should derive from the most recent annual production data that it is feasible to compile. The data should represent the mean of Chilean production, reflecting differing efficiencies from production at different ages and scales of production.

2.6. Data quality

Data quality is one of the main aspects in the construction of a national LCI database. For this project, an adapted version of the Pedigree Matrix (Weidema et al., 2012) will be used, evaluating the aspects in Table 4.

Table 4. Data quality indicators. Adapted from Weidema et al., 2012.

Indicator	**1**	**2**	**3**	**4**	**5**
Reliability	Verified data based on measurement	Verified data partly based on assumptions or non-verified data based on measurements	Non-verified data partly based on assumptions	Qualified estimate (e.g. by industry expert)	Non-qualified estimate or unknown origin
Completeness	Representative data from a sufficient sample of sites over an adequate period to even out normal fluctuations	Representative data from >50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from only some sites (<<50%) relevant for the market considered or >50% of sites but from shorter periods	Representative data from only one site relevant for the market considered or some sites but from shorter periods	Representativeness unknown or data from a small number of sites and from shorter periods
Temporal correlation	Less than 3 years of difference to the time period of the dataset	Less than 6 years of difference to the time period of the dataset	Less than 10 years of difference to the time period of the dataset	Less than 15 years of difference to the time period of the dataset	Age of data unknown or more than 15 years of difference to the time period of the data set
Geographical correlation	Data from area under study (e.g. O'Higgins)	Average data from larger area in which the study is included (e.g. Central Valley Region)	Data from area with similar conditions (e.g. Chile or Argentina)	Data from area with slightly similar conditions (e.g. South Africa)	Data from unknown or distinctly different area (North America in- stead of Middle East, OECD-Europe instead of Russia)
Further technological correlation	Data from enterprise, processes, and materials under study	Data from processes and materials under study (i.e. identical technology) but from different enterprises	Data from processes and materials under study but from different technology	Data on related processes or materials	Data on related processes or materials but different technology

The project will have no strict data quality goals, and the LCI will be as complete as possible, having in mind completeness over depth of them. Therefore, all data will be reported with their respective quality indicators. These indicators will be available for the users, so they can judge how useful the information is in order to use it.

2.7. Life cycle impact assessment

Impact assessment models that are used to interpret LCIA data are still emerging and evolving internationally. Since the early 1990s, numerous LCIA methodologies have been developed. The widespread use of several different methodologies creates confusion over which methodology to use, and criticism arises when the use of LCA gives different results depending on the methodology chosen (European Commission – Joint Research Centre 2008)

ISO 14044, section 4.4.2.2.3.c, recommends minimizing value-choices and assumptions during the selection of impact categories, category indicators and characterization models for the LCIA method, which is why this project includes a recommended LCIA method to be used.

The recommended LCIA method is proposed based on the research documented in:

- A Life Cycle Impact Assessment Method for Use in Australia – Classification, Characterization and Research Needs (Bengtsson et al. 2010):
- Identifying best existing practice for characterization modeling in life cycle impact assessment (Hauschild et al., 2013):

In essence the key considerations are:

- Whether the impact assessment is conducted at mid-point, end-point, or both.
- Whether the impact assessment should be based on best practice scientific methods per impact category (i.e. the BP LCI and AgAusLCI approach in Australia and the ILCD work in Hauschild et al., (2013)) or to adopt a ready-made LCIA method such as World Impact +, ReCiPe or CML.
- Which approach offers the most regionally appropriate impact assessment method.

The interim preference is to adopt ReCiPe on the basis that it:

- Has the flexibility to calculate impacts at both mid-point and end-point,
- Includes global characterization and normalization factors which are deemed by the project team to have higher degree of acceptance by the intended audience than European or US-centric LCIA methods.
- Aligns in terms of impact category selection with the best practice research for the BP LCI and ILCD
- Is actively maintained and updated.

The ReCiPe impact for water depletion and toxicity shall be replaced with the consumptive water use method developed by Ruidoutt and Pfister (2012) and USE Tox (Rosenbaum et al, 2008), respectively. The latter, including minor adaptations to Chile based on contribution analysis and known used substances.

Ionizing radiation will not be covered, as it was deemed not to be relevant for Chile, based on a contribution analysis made for avocado, table grapes, wine and chicken where it represented less than 1% of the normalized impact. In addition, there is no nuclear energy in Chile.

3. Summary and next steps

A data collection and life cycle impact assessment methodology has been developed for the Chilean LCI food and wine database project EcoBase, built upon international best practices and considering several stakeholders' inputs. The methodology will be submitted for critical review to achieve scientific validation before final publishing. This review will be by the Chilean LCA Network nationally and Quantis internationally, with the interest of seeking alignment with the World Food LCA Database.

It is expected that after the initial stage, these developed guidelines will allow more and better databases and products to be incorporated into the database.

Currently, the project is in data collection stage, following the methodology covered in this document. Secondary data collection has mostly been completed, while primary data collection is in progress. As mentioned in section 2.5, the data collection tool has been sent to several companies through trade organizations that are part of the project, and workshops to facilitate navigation and learning of the tool are being carried out with companies for most of the product categories. By mid-2014, data collection will be finalized, and by the end of the same year, a first draft of the LCI and calculators for the 16 selected product categories will be ready. After a critical revision by project partners, a final version of the LCI and the calculators will aim to be ready by early

2015 following which, the process of dissemination will begin until August 2015, which is when the project is expected to finalize.

It is important to mention that all project deliverables including methodology, life cycle inventories, calculators etc. will be publicly available on an online platform operated by project partners.

EcoBase Food is one step towards a more transparent and sustainable food sector in Chile, but there is still a long way to go in this direction. While currently the project includes 16 products, the aim is to expand the scope in future both in terms of increasing the number of products and industries involved, as well as improving the quality of the results as and when additional information and/or methods become available. Another future step currently being evaluated is better communication of these results to consumers, possibly under an eco-labelling program or a similar initiative.

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Integrating Nutritional Benefits and Impacts in a Life Cycle Assessment Framework: A US Dairy Consumption Case Study

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ABSTRACT

Although essential to understand the overall health impact of a food or diet, nutrition is not usually considered in food-related life cycle assessments (LCAs). As a case study to demonstrate comparing environmental and nutritional health impacts we investigate United States dairy consumption. Nutritional impacts, interpreted from disease burden epidemiology, are compared to health impacts from more traditional impacts (e.g. due to exposure to particulate matter emissions across the life cycle) considered in LCAs. After accounting for the present consumption, data relating dairy intake to public health suggest that low-fat milk leads to nutritional benefits up to one additional daily serving in the American diet. We demonstrate the importance of considering the whole-diet and nutritional trade-offs. The estimated health impacts of various dietary scenarios may be of comparable magnitude to environmental impacts suggesting the need for investigating the balance between dietary public health advantages and disadvantages in comparison to environmental impacts.

Keywords: dairy, dietary guidelines, LCA, nutrition

1. Introduction

Dietary guidelines, for instance suggested by the United States government, focus on improving health through dietary nutrition and do not consider environmental impacts occurring throughout the life cycle of the recommended diet (van Dooren et al., 2014). On the other hand, food-related Life Cycle Assessments (LCAs) assessing environmental and public health, tend to neglect nutritional advantages and disadvantages (Heller et al., 2013). In general, the assessment of emissions and nutrient intake is based on a functional unit relevant to the study question and the food system of interest. The selection of the functional unit is extremely important in LCA—specifically within food LCAs this point cannot be further emphasized, as it may change results when comparing two various food items or diets on different bases. Historically, there has been a large number of functional units based food LCAs (e.g. based on mass or volume of a single food type, quality corrected mass, a single nutrient, a full nutrient profile). Food oriented studies have mostly focused on environmental efficiency in crop or food production, for instance, how to decrease environmental impacts per unit output. (e.g. Garnett, 2013). In order to consider questions of impacts and benefits of dietary choices, i.e. consuming one food over another, assessment must be conducted in the context of the overall diet. These consumption oriented approaches (based e.g. on serving size, or full dietary or meal nutrients) are of growing interest as a means of informing consumer choices and policy decisions that affect dietary recommendations (e.g. Vieux et al, 2013).

Table 1 summarizes approaches to add nutritional impacts into environmental impact assessments the domain of application and limitations of each method, leading to the following statements: All present methods are impact oriented and do not enable a comparison of impacts with nutritional benefits. Several indices are themselves based on the contribution of the food to recommended nutrient intakes rather than based on epidemiological measures of health benefits and impacts. We expect in general when "quality" adjustments made including nutritional aspects, assessed impacts are reduced for items having positive nutritional benefits such as milk, fruit vegetables. This supports our thinking that if we can get to a better assessment of health benefits for products like milk then environmental impact in terms of expected negative human health impacts will be more balanced. However, trying to force various benefits and impacts associated to nutrition in the basis for comparison (the traditional functional unit) does not provide a clear view of the benefit of “positive” nutritional components versus the impact of “negative” nutritional components.

There is therefore a need to consider health benefits and impacts of nutrition in a parallel way to the latest quantitative development of the global burden of disease to incorporate diet-related risk factors, and consider Disability Adjusted Life Years as a basis for an improved index.

Table 1. Summary Table of Identified Approaches to Incorporating Nutrition in Environmental Impact Assessment

Approach	**Domain of application**	**Limitation / needs**	**Features of interest for framework**
a) Functional unit approaches			
Mass- or volume-based e.g. per kg or per liter	Internal (production oriented) assessments; identifying hotspots, evaluating abatement scenarios	Not appropriate for comparisons of different foods	Intermediary calculation step
Quality corrected mass e.g per kg FPCM	Normalizing product quality across different production practices; evaluating scenarios that affect product quality	Limited to simple nutritional quality assessments that can be related to a mass unit; not appropriate for comparison of disparate foods	Reconsideration of FPCM and impacts of allocated cream in overall diet scenarios
Single nutritional aspect, e.g. per g protein	Evaluating single nutritional dimensions across foods with related functions	Limited complexity; does not capture other nutritional differences	Impact and benefit of single nutrient could be estimated separately
Nutrient profile e.g. per index unit value	More complete nutritional picture; may be valuable in comparing foods	Negative impacts not adapted for functional unit; most current indices based on dietary recommendations rather than health outcome	Would be of interest to create a DALY weighted nutrient profile
Approach	**Domain of application**	**Limitation / needs**	**Features of interest for framework & next steps**
b) Consumption oriented approaches			
Serving size per serving size	Normalizes based on typical consumption quantities; can be used as a consumption-based unit for comparing individual food items, provided that health effects of nutrients are also considered	Does not link with nutrition or health; inconsistent definitions of servings	Testing scenario of one additional serving of milk
Diet-based for overall diet	Basis for consumption oriented assessment	Requires parallel assessment of nutritional quality/health; no basis to evaluate trade-offs	Basis for comparison framework at diet level
Equivalent nutrition diet for overall equivalent diet	Sound basis for comparing environmental impact of similar diets	Limited to nutritionally equivalent diets; definition of "nutritional equivalence" may be limited	Useful in considering isocaloric dietary substitution scenarios
Diet-level nutrient profile e.g. per index unit value of overall diet	May offer a comprehensive nutritional basis for diet comparisons	Negative impacts not adapted for functional unit; most current indices based on dietary recommendations rather than health outcome	Will be used as a comparative approach

To address these needs, we aim in this paper to 1) present a preliminary framework to show separately environmental and nutritional impacts together on a comparable scale, starting from serving size, 2) build on global burden of disease information and use of the DALY to compare human health impacts and benefits of nutrition, 3) provide a dairy case study to demonstrate the framework and analyze population-scale health effects of dietary changes.

2. Methods

2.1. Framework for comparing impacts and nutritional benefits of food

In an effort to bridge this research gap we build a preliminary framework and implement a dairy case study. Figure 1 schematically outlines the LCA framework to begin harmonizing nutritional and environmental impacts, where different food items within various diets may be associated with numerous environmental emission types which may influence population-scale health. Likewise, considered nutrients within those food items contribute to the nutrient index (i.e. nutrients consumed) and may also lead to various population-scale health impacts and benefits. To express human disease burden, many LCAs use the Disability Adjusted Life Year (DALY) (ILCD, 2010). Recently this metric has also been applied to quantify nutrition-based health impacts. For example, Murray et al. (2013) found that amongst a variety of health indicators and risk factors (e.g., smoking, violence, pollution etc.) inadequate diet generally accounts for 26% of deaths in the United States and 14% of DALYs. Benefiting from the implementation of the DALY metric in public health nutrition research and population-scale epidemiological evidence on minimum dietary risks, we build on work by Murray et al. (2013) and Lim et al. (2012) who performed large-scale epidemiological literature reviews on disease burden associated with specific food categories and nutrients, referring to both positive and negative health outcomes as nutritional impacts. Overall this approach offers an interesting possibility to compare nutritional benefits and impacts consistently with environmental impacts on human health.

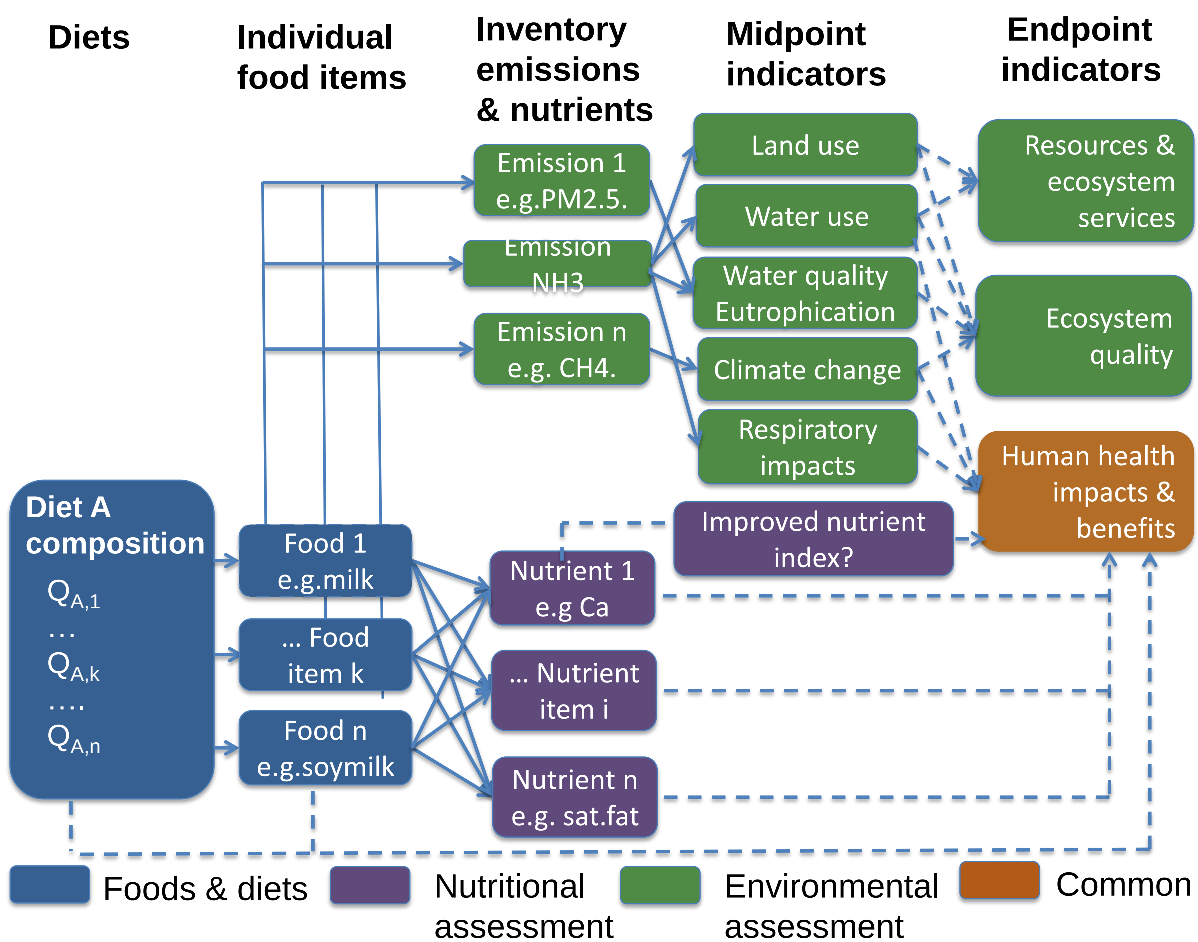

Figure 1. Framework to combine nutritional and environmental assessments. Dashed lines represent links that are useful to interpret midpoint categories, but whose quantification is also associated with a higher degree of uncertainty.

2.2. Dairy case study

a) Functional unit and scope

To test the LCA framework, which includes both nutrition and environmental impacts, we study a dairy case study. The Dietary Guidelines for Americans (USDA, 2010) recommends adult Americans consume 3 daily servings (≈740 g/day total) of low-fat or fat–free dairy products, close to doubling current consumption (USDA ERS. 2012). Previously performed LCAs (e.g. Thoma et al., 2013, Asselin-Balençon et al, 2013) have quantified various environmental impacts of dairy production in the US. Entirely missing from these LCAs, however, are the use phase nutritional impacts or benefits on human health due to dairy consumption. Building on the literature review from Heller et al., (2013), this research aims to incorporate nutrition related health effects of dairy into an LCA framework to compare them consistently with environmental impacts. We use the serving size as a common functional unit when combining nutritional and environmental impacts within LCA. This allows easy comparison with recommended dietary guidelines, based on serving sizes, and can also be easily converted to mass or volume, as well as to a corresponding nutritional intake or dietary function (e.g. calcium intake). We consider epidemiologically-based estimations of dairy's contribution to the global burden of disease and caloric intake to evaluate the health impacts due to nutritional components. For environmental impact, we focus on greenhouse gas (GHG) emissions and human health impacts of particulate matter (PM). We select these two impact categories because of their high relevance, and they have been extensively studied in relation to the dairy industry by Thoma et al. 2013 and Henderson et al. 2013. Additionally, human health effects of PM have also been assessed through various epidemiological studies (Pelucchi et al., 2009) and can thus be compared with nutritional impacts.

b) Defining the American diet and dietary context

In order to understand how a shift in dairy consumption may affect overall nutritional intake, we must first define the current "average" diet in the US. We assess two available datasets: The National Health and Nutrition Examination Survey (NHANES) and the USDA-maintained Loss Adjusted Food Availability (LAFA) data series (USDA ERS 2012). Survey-based approaches, like NHANES, are known to under-report food intakes; they are also comprised of as-consumed (e.g., processed) food items, with no consideration of the life cycle of a food item e.g. from the commodity or ingredient level. LAFA, on the other hand, measures the use of food commodities (e.g., wheat) by tracking the US marketplace. Generally, the available supply is the difference between the sum of production, imports and beginning stocks, and the sum of non-food use (e.g. industrial uses), exports, and ending stocks for a given calendar year. In the LAFA series, the available supplies for over two hundred commodities are adjusted by percent loss assumptions, for instance plate waste and food spoilage in order to account for various processes effecting food loss along the chain farm to retail. Availability adjusted for loss can serve as a useful proxy for per capita food consumption in the US. While LAFA data is presented at the food commodity level (i.e., raw farm products like wheat and corn rather than consumables like bread or tortilla chips), this level is far more manageable from the environmental impact perspective; LAFA data also allows to account for supply chain losses, which contribute to environmental impact but are not consumed (and therefore do not contribute to nutritional health effects).Therefore, we propose using a recent 2010 LAFA data series to define average US food consumption (the background average US diet) within this proof-of-concept case study. These data are available both on a weight and on a food pattern equivalent basis, permitting consistent connection between environmental impact and nutritional data.

c) Scenarios

In general, it is important to consider the nutritional and health effects from consumption of a single food item within the full dietary context. From average diet information we find that the average US adult diet includes 1.5 cup equivalents (with around 245 grams of milk per cup) of dairy consumed per person per day, about half the recommended value. To investigate the shift from average consumption to recommended consumption we evaluate three dietary scenarios. These are hypothetical examples to implement this framework; unfortunately there is limited data available to predict actual dietary changes. Starting from the average diet (scenario 0) at population-scale we investigate the effects if people generally will (Figure 2):

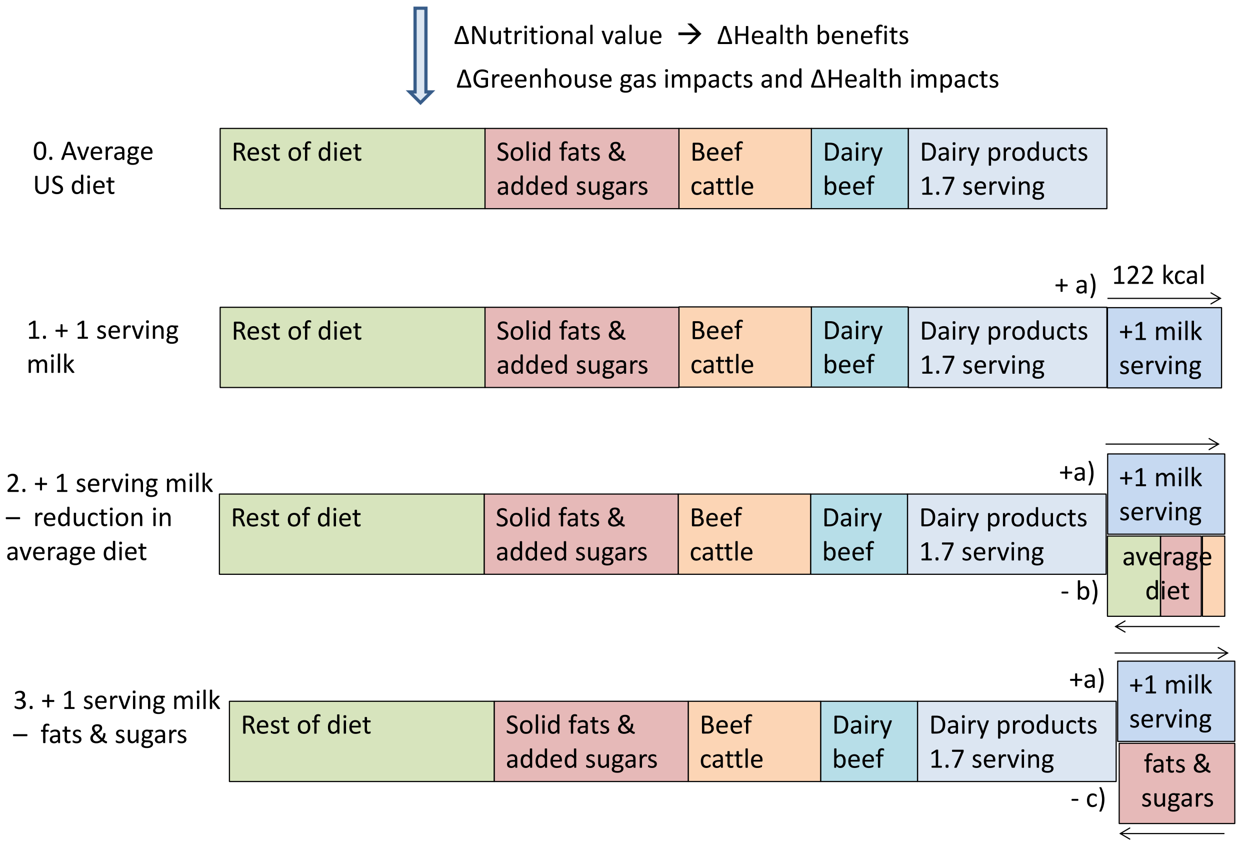

Figure 2. Scenario schematic: Changes in nutritional/health benefits and environmental impacts (greenhouse gas, example) associated with one additional serving of milk within the diet.

1. Add an additional serving (1 cup) of "2% milk," that is fluid milk containing 2% fat, (per capita per day) to the diet, with no change to the rest of the diet; i.e., this scenario would result in an increased caloric intake over the average diet baseline.
2. Add an additional serving of 2% milk while subtracting an equal caloric quantity from the overall average diet. Thus, the 1 cup of milk would be added to a diet that is proportionally reduced by ~122 kcals (the energy content in 1 cup of 2% milk). The resulting diet would be iso-caloric with the average diet baseline.
3. Add an additional 1 cup of 2% milk while subtracting an equal caloric quantity of "empty calorie" foods. This will likely be a proportional representation from the Dietary Guidelines' "solid fats and added sugars" category. The resulting diet would be iso-caloric with the average diet baseline.

We use these specific scenario decisions based on the following logic: In response to dissemination of dietary guidelines, consumers may acknowledge the governmental recommendation for increasing lower-fat dairy consumption, and chose to supplement their diet with a low-fat milk product (here considered 2% fat, because it is the most commonly consumed lower fat form of milk). This dietary change: 1) may not result in compensatory removal of equivalent calories from the rest of the diet (e.g. Duffy and Popkin 2007); 2) may result in compensatory removal of calories resulting in an a diet approximately iso-caloric to before dairy supplementation; 3) may result in removal of other beverages such as sodas from the diet (e.g. Cavadini et al. 2000).

3. Results compilation and case study

3.1. Nutrition-oriented assessment and relation to health

To assess nutrition-related health impacts, we use a global burden of disease (GBD) approach, building off of Murray et al. (2013) and Lim et al. (2012). Specifically, we build on the risk ratios from the GBD sources and minimum risk thresholds to extend nutrition-oriented work that associates nutritional/dietary benefits and risks with DALYs.

The benefits associated with increasing dairy consumption are dependent on the status-quo consumption or dairy products. Within the GBD risk framework, there are no additional health benefits to increase servings above the consumption threshold corresponding to the reported *minimum risk level*. Given the adult population (19+ yrs) in NHANES 2009-2010 had a reported milk intake of 0.87 cup per day (unpublished data), we can calculate the increase in amount of milk to reach the milk consumption threshold corresponding to the minimal risk level according to Lim, et al (2012). Assuming we standardize milk to 2% reduced fat milk with 236 g/cup, we estimate that currently American adults are consuming 205 g milk/day (the product of 0.87 cup and 236 g/cup). With a reported minimum risk threshold level of 450 g/day, we can deduce that adult Americans would need to consume *an additional* 245 g milk per day on average, which is about 1.04 cups (245 g milk/236 g per cup), to reach the minimal risk level outlined in Lim et al., (2012).

As a comparative exploration, we perform the same calculation based on nutrients instead of dairy or milk as a whole-food. The reported minimal risk for calcium intake is 1200 mg/day. Assuming a non-dairy calcium intake of about 385 mg/day (Fulgoni et al. 2011) and calcium from dairy as 516 mg calcium (1.72 cup equivalents of all dairy products from NHANES 2009-2010 with each cup equivalent having approximately 300 mg of calcium) we deduce we need an additional 299 mg calcium/day (1200-385-516=299) to achieve the minimal risk level outlined in Lim et al., (2012). Again, this corresponds to about one cup of 2% milk.

It appears Murray, et al., (2013) used ounces of weight rather than fluid ounces for their calculation for impact of milk (226.8g/8 oz=28.35g). To keep units similar to relative risks we can perform similar calculations; based on these relative risks, the change in milk in cup equivalents is 0.96 cup eq of 2% milk, which again corresponds to about one cup.

These different preliminary calculations suggest that an increase of *one serving of milk* will maximize health benefits for the primary dairy-related risk categories. Increases above this level would increase impacts without additional benefits in these categories. Further research will be needed to define minimal risks for other health categories (e.g., bone health, blood pressure).

3.2. Quantification of greenhouse gas emissions

Detailed results from the Innovation Center sponsored milk LCA studies (Thoma et al. 2013; Henderson et al. 2013) are used to estimate greenhouse gas emissions from the life cycle of fluid milk production. Additionally, to broadly estimate GHG emissions from the general diet we generate a dataset of 250 data points aggregated from a variety of published studies on greenhouse gas emissions from the production of specific foods in specific places. Where multiple emission factors exist in the dataset for the same food, an average is taken. For foods without specific data points, proxies are formed by averaging related foods. Several limitations exist in using this dataset: many studies are not specific to US production scenarios; significant variability can exist between studies of the same food; transport and other variables are not treated in a consistent fashion across studies. Still, the dataset is useful for a pilot level study; it captures most of the important foods in the LAFA dataset and is the most reasonable collection of data current available.

3.3. Quantification of Particulate Matter (respiratory inorganics) emissions and related impacts

The largest contributors in typical food and agricultural systems to respiratory inorganic impacts are particulate matter (PM) from internal combustion engines (i.e., tractor use, transportation) and ammonia emissions (a PM precursor), primarily from manure storage and handling. The Comprehensive LCA of Fluid Milk (Henderson et al. 2013) models these emissions (and impacts) related to PM for the US milk production chain, and will thus be used in this case study. Since most published LCA studies on foods do not report these emissions or impacts, we propose the following approach to provide a first order estimate of the PM associated with producing the average US diet. The majority of PM from food systems comes from direct ammonia emissions during agricultural production (generating secondary PM) and from tractor usage and from transportation (primary and secondary PM). A preliminary study carried out on the 4000 Ecoinvent processes shows that PM from all sectors can be interpolated from correlations between PM and greenhouse gas emissions (GHGE), with specific coefficients for agriculture production and for transportation. We estimate the proportion of GHGE from the production of individual food types using studies that report the contribution of field emissions, tractors and transportation to various food group GHGE (Weber and Matthews 2008; Hoolohan et al. 2013; Meier and Christen 2013). These studies will be combined with the Ecoinvent derived correlation to estimate direct emissions of PM2.5 (so-called primary PM) as well as emissions of precursors (NH3, NOx, SO2) of secondary PM (this important PM precursor for the average diet.

As described by Fantke et al. (2014), a starting point to determine the human health impacts associated with primary and secondary particulate is the effort of an earlier UNEP/SETAC working group that has designed a framework and proposed a set of default human intake fractions (iF) associated with PM emissions for use in LCIA (Humbert et al. 2011). This effort was consistent with other consensus building actions in the frame of comparative chemical assessments. However, this effort is limited to the part of the impact pathway from emissions to concentration and human intake, but does not cover the part from human intake to finally health effects. This will be complemented according to the latest recommendations of the LCIA guidance task force for PM related health impacts based on dose-response from the 2010 global burden of disease (Lim et al. 2012; Murray et al. 2013) as well as calculations based on the American Cancer Society Study of Particulate Air Pollution and Mortality (Gronlund et al. 2013; Pope III et al. 2002). Since nutrition impacts and benefits will also be mostly based on the 2010 burden of disease, this will ensure a good consistency between environmental impacts and nutrition.

3.4. Overall trade-offs

With considering only one serving additional in each dietary scenario, preliminary results show the following tendencies. The increase in calories may dominate, generating important additional impacts. In the iso-caloric scenario 2, the nutritional benefit of milk becomes important and will counter balance the human health impacts of PM over the life cycle. Dietary scenario 3 provides the most substantial nutritional advantage by also reducing the nutritional impacts of added sugar. We finally analyze the trade-offs between human health and global warming impacts for each scenario.

4. Discussion

Considering the present US consumption, and limited epidemiological data on disease burden, our results suggests that lower-fat milk leads to nutritional benefits up to one additional serving. Nutrition is a complex issue and we do not attempt to cover all intricacies in this case study. For example we do not consider non-nutritional health impacts such as social/cultural issues with dietary changes, or other health related issues such as bowel motility or indigestion related to lactose intolerance. We also do not consider possible variations in health impacts for different age groups (i.e. a health advantage of consuming dairy may be more measurable for children and teenagers than for elderly populations) and consider the general order of magnitude of potential impacts at the population scale and not the health impacts for a given individual. We also do not consider at this stage other potential benefits of dairy products on other risk areas such as bone health and blood pressure, whereas the Lim et al. (2012) and Murray et al (2013) studies consider only carcinogenic risks (e.g. balancing associations of dairy consumption with decreased risk of colorectal cancer and increased risk of prostate cancer).

It is important as population grows and food and environmental resources become strained that research continues on harmonizing nutritional and environmental public health impacts of foods and diets. This study assesses epidemiological data on disease burden in relation to one food group (dairy) and one associated nutrient (calcium) in the context of governmental dietary regulations and LCA. This type of analysis may help regulators in selection of dietary guidelines and promote holistic consideration of both the environment and the direct nutritional implications. Our findings emphasize a clear need for more research on dietary choices and for understanding whether or not shifts in the diet are compensated for by other food groups. Generally the framework we present provides the groundwork for considering both the nutritional and environmental impacts of a specific food, and for also considering the entire dietary context, which we find essential.

5. Conclusion

In all, this work suggests that in order to optimize public health dietary recommendations, nutritional impacts and benefits should be considered with respect to the entire diet as well as the food's life cycle. This work provides a stepping-stone for consistently incorporating nutritional effects into an LCA framework.

Accounting for the nutritional function of food is a perennial problem in assessing the environmental impact of food production in the LCA framework. The review presented here outlines a functional unit approach that also includes dietary context. We propose a novel framework for bringing environmental and nutritional aspects of food together using the global burden of disease approach to evaluating risk factors using the DALY metric. This approach allows expression of behavioral choices such as diet composition in terms of the associated disease risks. Under such a framework, nutritional and environmental health impacts of a dietary change may be evaluated in equivalent units, permitting a quantitative estimate of the complements and trade-offs between nutrition and environment. Regarding the limited epidemiological data relating food items to possible health impacts, these preliminary investigations do not offer exact calculations or the health risks for a given individual, but instead should be considered as an indication of the importance of balancing nutritional and environmental health impacts, as these may lead to contradicting conclusions depending on the population-scale dietary context.

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Incorporating Health Impacts from Exposure to Chemicals in Food Packaging in LCA

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ABSTRACT

Life cycle assessments (LCA) on the environmental and public health impacts of food and beverage packaging materials have found some advantages to plastic over glass. Entirely missing from these evaluations are the health impacts of possible chemical, e.g. endocrine disruptor, exposure through migration of chemicals from the packaging into the food product. We build a framework based on a life cycle perspective to predict which chemicals may be in a package that are not intentionally added ingredients, and we apply this approach to the US EPA's CPCAT database. In total we find 1,154 chemicals within the CPCAT database related to food-contact materials; out of these 107 are potential endocrine disruptors according to the TEDX list of endocrine disruptors. We also build a framework in an effort to begin harmonizing LCA to include health impacts of chemical exposure related to food packaging in conjunction with other traditional LCA environmental impact categories.

Keywords: Endocrine disruptor, exposure, food packaging, LCA

1. Introduction

Life cycle assessment (LCA) has long guided industry in selection of food and beverage packaging materials to minimize environmental and human health impacts (Curran 1993; Humbert et al. 2009; Marsh et al. 2007; Roy et al. 2009). Often, results suggest that plastic packaging has marginally better performance than glass, for instance for baby food containers (Humbert et al. 2009) or for alcoholic beverages (Smith, 2009). Packaging efficacy and chemical emissions during transport and disposal have been the focal issues in these LCAs. Although human health may be a relevant impact category in such LCAs, for instance due to inhalation of particulate matter or organic chemicals, lacking from these impact assessments is addressing the possibility of chemicals leaching from packaging into the ingestible products. Contamination of food products from packaging has been documented, and of particular concern are plastics and paper treated with fluorinated chemicals (Loyo-Rosales et al. 2004; Muncke et al. 2011; Tittlemier et al. 2007; Trier et al. 2011; Wagner et al. 2009). Some fluorinated and plastic-associated chemicals have demonstrated endocrine disrupting properties, meaning they may interfere with various natural hormone systems within the human body (Muncke et al. 2014; Zoeller et al. 2012). Endocrine disruptors are a chemical category of high exposure concern and are particularly complex to assess because of their atypical toxicology, often with non-linear dose-response curves and substantial low-dose effects, as well as various effects in exposed populations due to interference with hormone-dependent human development and functioning (Colborn et al. 1993; Vandenberg et al. 2012). To our knowledge, exposure to endocrine disruptors through ingestion of packaged foods has not been addressed in LCAs. In general, LCAs evaluating health impacts of consumer products, even food products, focus on health impacts due to exposure to and intake of environmental pollutants and health impacts due to chemicals embodied within the product are neglected.

Empirical evidence suggests that food contact materials, such as packaging, lead to human exposure to chemicals of concern (Loyo-Rosales et al. 2004; Tittlemier et al. 2007; Wagner et al. 2009). For example, significantly lower levels of potentially endocrine disrupting chemicals bisphenol A and bis(2-ethyhexyl) phthalate were observed in the urine of participants while undergoing a "fresh food diet intervention," where packaged foods were limited (Rudel et al. 2011). In all, substantial evidence suggests humans are exposed to chemicals within food packaging, and thus this exposure avenue must be considered in LCAs of food packaging materials in order to provide a holistic assessment of product impacts.

Risk assessment is a parallel field of study to LCA which also offers insight regarding the exposure to and human health risks of chemicals contained in e.g. consumer products. For instance, risk assessments of chemicals within consumer products tend to focus on risks associated with the product's ingredients (Wormuth et al. 2007). Although considering ingredients is an essential first step, this strategy neglects chemicals which may be in the food packaging product that are not added intentionally as a final ingredient (Muncke et al. 2014). Thus, whether product-associated health risks are considered from a risk or LCA perspective, considering the general life cycle perspective of a food package offers a more holistic consideration of what chemicals may end up in the food due to packaging that were not intentionally added package ingredients.

The present study aims to *provide a framework to incorporate toxicological health impacts due to exposure to chemicals in food packaging* that is harmonized with established LCIA frameworks used to calculate human toxicological health impacts due to life cycle environmental emissions. We consider the life cycle perspective in two distinct ways: 1) the life cycle perspective is applied in an effort to determine all possible avenues of chemical incorporation into a food package and thus into a food product and 2) the life cycle perspective is applied as an overall framework to calculate human health impacts which may result from different stages of the food product's existence.

2. Framework development

2.1. Life cycle consideration to identify chemical constituents

Literature review suggests human ingestion of chemicals due to food packaging may be influenced from various life cycle stages and through various avenues (Figure 1.). These influences were split into two distinct sequences representing before packaging and after and during packaging. For example, before a product was distributed to consumers, chemicals may be incorporated into food packaging because they: were the result of a chemical reaction or incomplete chemical reaction occurring during material acquisition or manufacturing (Muncke et al. 2014), they were ingredients of the raw materials or ingredients required for manufacturing (Lau et al. 2000; Tittlemier et al. 2007; Trier et al. 2011), or they were incorporated incidentally during manufacturing through contamination from equipment or the environment, as for instance is observed in other consumer products such as tampons (FDA, 2013). Likewise, during and after the packaging process, that is while and after the food was enclosed by the package, although contamination of the packaging material may now be negligible, migration or diffusion of chemicals from packaging into the food may occur. Migration may occur passively through the passage of time depending on temperature, pressure, and the physicochemical interactions of the food-package combination. Additionally, there are various processes which require further consideration. For example, food pasteurization through thermal sterilization often occurs within the packaging (Ranken et al. 1997), or a consumer may microwave the food within the package (Begley et al. 2005); these processes may increase migration of chemicals into the food item or cause a heat-dependent chemical reaction.

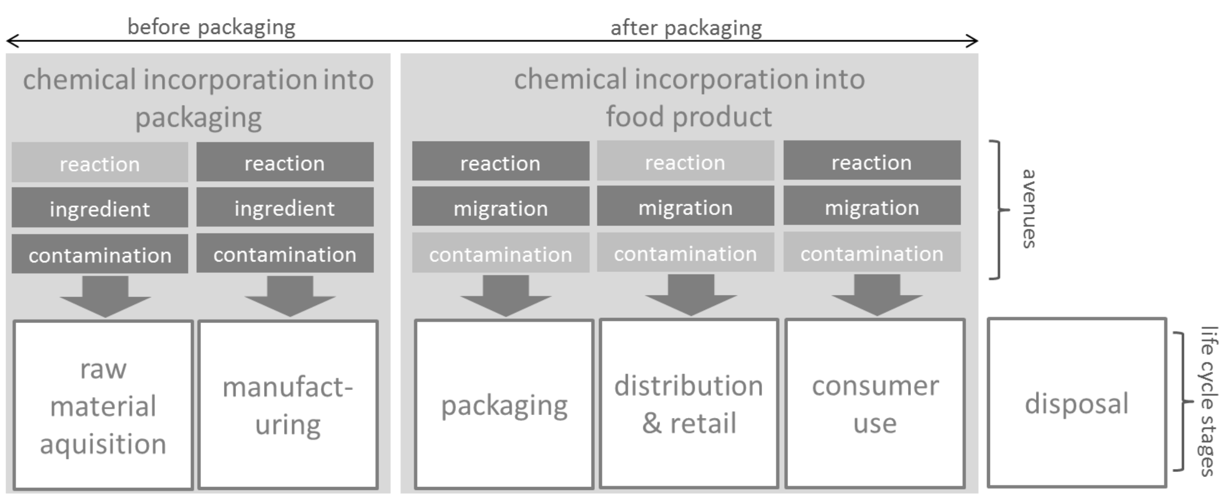

Figure 1. Relevant stages of a packaged food product life cycle, where (1) chemicals may be incorporated into the package, and (2) where these chemicals may be subsequently incorporated into the food product by various avenues (light shaded avenues are considered less relevant).

To obtain preliminary information from a life cycle perspective on which chemicals may be within food via food packaging, we mined the newly developed consumer product category (CPCAT) database publically available through the US EPA (Dionisio et al. 2014, in preparation) to look at chemicals known to be in food contact materials, as well as those known to be used in manufacturing, and as raw materials.

2.2. Development of a framework to incorporate chemical exposure to food packaging in LCA

In our framework development we considered that LCAs which include human health impacts of food packaging products may be performed using an ISO framework, generally with the following phases: 1) defining the goal and scope of the study and relevant life cycle stages; 2) collection and analysis of inventory, that is inputs and outputs of the product system; 3) assessment of impacts associated with outputs of the product system; and 4) interpretation of impacts. Typical relevant stages of a product's life cycle include, but are not limited to: the acquisition of raw materials, manufacturing, distribution, use of product, and disposal of the product. Throughout and between these life cycle stages there may be various processes also contributing to emissions such as due to transportation and waste streams.

In this study, we defined the goal and the scope around the need to include human exposure to chemicals within food packaging into an overall LCA framework in order to holistically quantify associated human health impacts. We therefore defined the relevant life cycle stages required to model this process slightly differently from traditional LCA life cycle stages. To summarize from Section 2.1, chemicals may end up in a food product via the packaging through acquired raw materials, while manufactured, while the actual packaging occurred, while the food was distributed and retailed, and while the consumer used the food product (Figure 2, top row of boxes and solid lines).

Likewise, each of these steps may lead to environmental emissions due to for example waste streams or energy use; further the disposal of the food packaging must also be considered to evaluate environmental emissions (Figure 2, top row of boxes and dotted lines). Disposal may also be important to evaluating the population-scale intake of chemicals within foods due to packaging, if food waste is significant. Inventory analysis needs to account for environmental emissions throughout the life cycle of the product, for instance from raw material acquisition to the disposal and any transportation processes occurring between and within different stages. Using the CPCAT database, a parallel inventorying process was performed analysing the various chemical inputs resulting from distinct life cycle stages in order to predict which chemicals and to what extent they may end up in a packaged food product.

In order to assess the impacts of these parallel inventories, exposure and intake of chemicals must be assessed. The intake fraction, iF, concept (Bennett et al. 2002) is often employed in LCIA models assessing human-toxicological impacts from chemical. Likewise, the product intake fraction, PiF, (Jolliet et al. 2014 in preparation; Ernstoff et al. 2014 in preparation) is a concept proposed for use in LCIA to estimate population-scale and product-user intake of substances contained in the product. The end goal was to assess the overall human health impact from exposure to chemicals in food packaging materials. This required consistent consideration of the health impacts due to environmental emissions (i.e. using the iF) versus health impacts due to use stage exposure and intake of chemicals in the food package migrating into the food product (i.e. using the PiF).

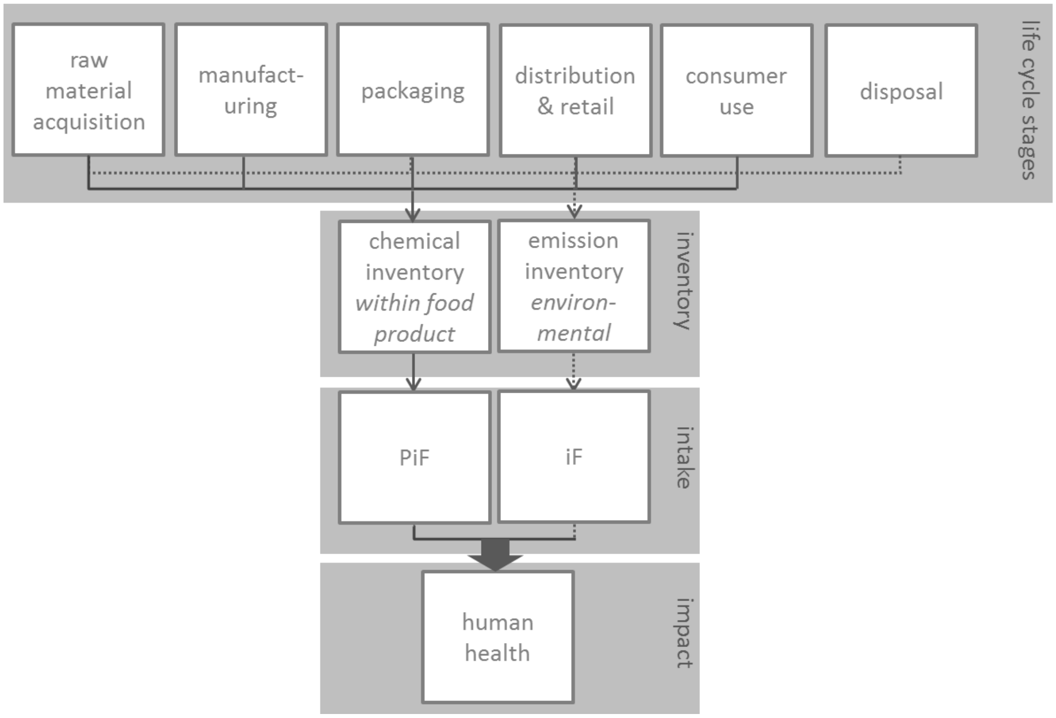

Figure 2. Contribution of relevant stages of the package's life cycle to human health impacts. Solid lines correspond to avenues in which chemicals may be incorporated into a food product, leading to human exposure and intake calculated as a product intake fraction (PiF) and dotted lines correspond to avenues in which chemicals may be emitted to the environment, leading to subsequent human exposure and intake calculated as an intake fraction (iF).

2.3.Issue of endocrine disruption

Many chemicals used within food packaging are known or suspected endocrine disruptors (Muncke et al. 2014). Perhaps due to their complicated and atypical toxicology (e.g. non-linear and non-monotonic dose response curves) (Vandenberg et al. 2012), to our knowledge, there are no methods to incorporate endocrine disruption toxicity into an LCIA framework for human-toxicological impacts, which currently relies on the assumption of a linear dose response (Rosenbaum et al. 2011). In an effort to begin including endocrine disruptors into LCIA we first employed a flagging method where chemicals were flagged if they are on lists of suspected endocrine disruptors (e.g. TEDX 2014). From here we investigated EPA approaches to approximate low-dose nonlinear chemical exposures, where at the population scale, due to tremendous individual variation, a near linear dose-response may be observed, Figure 3. (US EPA 2009).

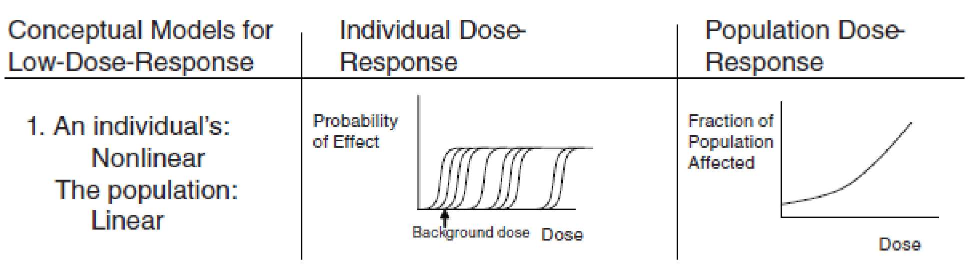

Figure 3. Demonstration of possible population scale linear dose response even when an individual's dose response may be nonlinear, taken from US EPA (2009).

3. Results

The main purpose of this paper was to develop a framework to consider exposure to chemicals within food packaging, as well as to better assess which chemicals may be incorporated into a food package and subsequently into a food product along the supply chain, therefore the main results were demonstrated in the framework development section above. Additionally, we mined the newly released US EPA consumer product category (CPCAT) database to begin understanding which chemicals may be in food products due to food packaging. In total we find 1,154 chemicals within the CPCAT database related to food-contact materials; out of these 107 are potential endocrine disruptors according to the TEDX list of endocrine disruptors. When distinguishing by life cycle stage, we find 235 chemicals associated with use as *raw materials* for food-contact products, such as packaging; 52 of these are potential endocrine disruptors. For the *manufacturing* life cycle stage we find 12 chemicals, 4 of which are potential endocrine disruptors. Table 1 shows all the manufacturing related chemicals and a sub-sample of the raw-materials related chemicals. When looking to understand the chemicals in a food package which may not be listed ingredients, raw material and manufacturing labels within CPCAT are the two distinguishable life cycle stages. All the other chemicals associated to food contact materials within the CPCAT database are listed in the database associated with the given product and thus are likely intentionally added ingredients.

Table 1. Example of chemicals likely used in food packaging manufacturing as they are listed in the CPCAT database as chemicals approved for manufacturing of food-contact materials.

Life cycle stage	CAS RN	Chemical Name	Potential endocrine disruptors
Raw material	106-89-8	1-chloro-2,3-epoxypropane	X
Raw material	126-99-8	2-chlorobuta-1,3-diene	X
Raw material	149-57-5	2-ethyl-1-hexanoic acid	X
Manufacturing	63148-62-9	baysilon	
Manufacturing	71-43-2	benzene	X
Raw material	117-81-7	bis(2-ethylhexyl) phthalate	X
Raw material	80-05-7	bisphenol A	X
Manufacturing	10043-35-3	boric acid	X
Raw material	84-74-2	dibutyl phthalate	X
Raw material	75-21-8	ethylene oxide	X
Raw material	96-45-7	imidazolidine-2-thione	X
Manufacturing	7664-38-2	orthophosphoric acid	
Raw material	131-57-7	oxybenzone	X
Manufacturing	25322-68-3	polyox WSR-N 60	
Raw material	25013-16-5	tert-butyl-4-methoxyphenol	X
Manufacturing	13463-67-7	titanium dioxide	
Manufacturing	108-88-3	toluene	X
Manufacturing	1330-20-7	xylene	X
Raw material	137-30-4	ziram	X

4. Discussion

The development of this framework is an iterative process and further insight will be gained for example, when case studies are implemented. In this discussion we focus on issues related to understanding chemical exposures due to food packaging, and do not address uncertainties related to the traditional LCIA approaches regarding environmental emission inventory of toxic chemicals.

It is difficult to identify the vast number of chemicals which may end up in a food product due to packaging because of the various contributions of the life cycle stages, and the CPCAT database is certainly not exhaustive. Further, once the chemical inventory is created and the majority of chemicals within food packaging are identified, large uncertainties are expected when estimating the subsequent concentration or migration potential of each chemical from the packaging into the food. A large amount of data is required on the variables and processes which may influence migration, for example on packaging strategies (e.g. if foods are pasteurized within the packaging), time and average storage temperature during the distribution and retail, and consumer use (e.g. if foods are microwaved within the packaging).

Further, even in the cases when the concentration of a chemical contaminant in the end food product is known perhaps due to analytical detection, data will be needed on population scale ingestion of the food product, and of course toxicity or dose response in the case of estimating resulting health impacts. Specifically for endocrine disruptors, where new information is being generated at a fast rate, we expect many years of research required to provide methods to consistently include such impacts within an LCA framework.

Aside from uncertainties in our analysis and within the defined system boundaries, there are also various considerations outside the scope of this study which may also influence the overall health impact related to a food package. For example, the food package's efficacy in controlling the growth of pathogenic bacteria on food items is an important contribution to human health.

5. Conclusion

In all, we suspect it is essential to include chemical exposures via food packaging in order to estimate the overall health impact of a food product. Many chemicals associated with food contact materials are potential endocrine disruptors and are therefore of particular concern. Insight gained through the life cycle perspective may help identify chemicals of concern which are not intentionally added ingredients, and also help consistently evaluate the extent of a food package's effect on public health both mitigated through environmental emissions and exposure occurring through consuming the packaged food. In this paper we provide the ground-work to advance future work addressing these necessary research gaps and a framework which must be iteratively developed with advances in knowledge.

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Indirect Land Use Change and GHG emissions of two biodiesel pathways in Spain

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ABSTRACT

Imported biodiesel has accounted for a large share of the total amount consumed in Spain, the main supplier of which was Argentina at least until anti-dumping duties on biodiesel imports from this origin were approved by the European Commission in November 2013. A consequential LCA is carried out in the present study to compare this pathway, which was the prevailing one until almost 2014, with the alternative of using domestic biodiesel from Used Cooking Oil (UCO). System expansion is performed in order to take the indirect functions of both systems into account, functions arising from interactions between co-products (protein meals) in the animal feed market. The marginal suppliers of these co-products in the international market are identified and emissions from direct and indirect Land Use Change (LUC) are calculated. When they are not considered, imported soybean biodiesel leads to lower GHG emissions, due to the carbon uptake by biomass. However, when global LUC is taken into account, UCO biodiesel generates a much lower impact, because it causes a contraction in the area diverted to biofuel feedstock production in other parts of the world. The results underline the importance of considering emissions from LUC when comparing biodiesel alternatives and, thus, interactions in the global market must be addressed.

Keywords: biodiesel, consequential LCA, greenhouse gas, land use change, Used Cooking Oil

1. Introduction

Over the last ten years, the worldwide consumption of biofuels has been on the rise for both economic and political reasons. Together with rising oil prices, public policies have been instrumental in promoting the use of non-petroleum based fuels in countries such as the United States (US), Brazil or the European Union (EU). Specifically, European Directive 2009/28/CE aims to reduce the greenhouse gas (GHG) emissions by introducing a blending mandate of 10% biofuel share in the motor fuel market of the Member States by 2020. However, despite the increased demand for biodiesel in Spain as a consequence of this Directive, the biodiesel sector is currently working at around 10% of its total production capacity. While the renewable energy target in transport, which was 6.5%, was nearly reached in 2012, in that very year imported biodiesel accounted for 79% of the market share (Datacomex 2013). Most of this biodiesel was imported from Argentina, providing almost half of the total consumed.

In addition, this Directive sets out a sustainability criterion requiring biofuels to emit at least 35% less GHG than the replaced fossil fuel; emissions must be calculated over the entire life cycle and must include the emissions from land conversion necessary for the cultivation of the raw materials. This phenomenon, known as Land Use Change (LUC), refers to potential changes in the carbon stock of the soil and biomass that take place when the land is diverted to biofuel production. Apart from the direct LUC (dLUC) as a consequence of an increased demand for biofuels, indirect LUC (iLUC) takes place through global market-mediated effects, prompted by the displacement of existing productive uses. In an attempt to promote the production of biofuels which do not lead to substantial losses of land carbon stock either in the producing region or elsewhere (known as *advanced* biofuels), the EU recently launched another proposal, COM 595 (European Commission, 2012), still under debate. It seeks to minimize GHG emissions from iLUC by limiting the contribution of conventional biofuels to that 10% target, and obliging Member States to report the estimated iLUC emissions from the biofuels they produce.

Against this background, the production of biodiesel from Used Cooking Oil (UCO) in Spain may help to increase the energy independence and boost the national industry, given the requirements imposed by COM 595. In fact, UCO was the second most commonly used feedstock for producing domestic biodiesel in 2011 (CNE 2013). Hence, it constitutes a viable alternative to the importing of biofuels, especially to those coming from feedstock used in the food and feed markets (known as *first generation* biofuels). The aim of this study is to compare overall GHG emissions –including those from iLUC– of two current pathways for biodiesel consumption in Spain: importing soybean biodiesel from Argentina and producing biodiesel from UCO. It must be taken into account though that the imports of biodiesel from Argentina have markedly declined during the first few months of 2014, as a consequence of the anti-dumping duties which have recently been approved by the European Commission (Regulation 490/2013). However, most of the biodiesel is currently produced from imported feedstock, and imports of vegetable oil into the EU have been increasing sharply. Soybean oil from Argentina accounts for a remarkable market share in Spain, only surpassed by palm oil imports from Malaysia and Indonesia. Therefore, this pathway for biodiesel consumption still has iLUC effects, similar to those arising from importing the manufactured product, with the only difference that now the transesterification takes place in Spain.

2. Methods

2.1. System description

A consequential approach is used to perform LCA according to the ISO standards (ISO 2006a, b); this is because crop displacement is the result of interactions among global agricultural markets and, as such, this is the way to quantify these future indirect responses. Economic modeling is a very useful tool with which to quantify these effects, and it has been used by authors such as Banse et al. (2010), Hertel et al. (2010) or Kløverpris et al. (2008) in the field of bioenergy. However, some practitioners have developed a methodology for quantifying the environmental consequences of increased biofuel consumption within the LCA framework (Dalgaard et al. 2008; Reinhard and Zah 2009, 2011; Schmidt 2010; Schmidt and Weidema 2008). It is based on system expansion, in order also to include interactions due to biofuel co-products such as protein meals, which may fulfill different functions in other markets. These studies are built on causal relationships, which model the substitution effects between those *marginal suppliers* affected by changes in the production system. The marginal supplier of a co-product is defined by Weidema (2003) as the most competitive in the international market. For each co-product, the marginal supplier of the substitute must be identified.

According to the procedure described by Weidema (2003), two scenarios are defined: Scenario 1, where soybean biodiesel is imported from Argentina, and Scenario 2, where it is produced from UCO collected in Spain. The functional unit is 1 MJ of biodiesel in regional storage in Spain. Scenario 1 mainly consists of *soybean farming*, *soybean oil extraction* and the subsequent oil refining, *soybean methyl-ester (ME) production* by transesterification and *soybean ME export to Spain* by tanker. Scenario 2 includes *UCO collection*, *UCO-ME production* and *transport within Spain*. Furthermore, due to market responses, a loop between co-products is identified and iterated against 0 to estimate the indirect effects of 1 additional MJ of biodiesel consumed in Spain, following the same principles used by Dalgaard et al. (2008) and Reinhard and Zah (2009).

In Scenario 1, it is assumed that increasing the production of soybean ME in Argentina to meet the Spanish demand reduces the amount of soybean oil in the international market, the shortage of which has to be compensated for by Malaysia as the marginal supplier of vegetable oil. According to the *ceteris paribus* assumption (Ekvall 2000), the demand for Malaysian oil expands 1:1, with the subsequent expansion in the agricultural land. As side effects, palm kernel meal production increases too, affecting the demand for Brazilian soybean meal in the animal feed market. The loop between the soybean meal and the soybean oil leads to a net increase in the production of palm oil in Malaysia (+25.27 g), and to a decrease in the production of soybean meal in Brazil (-3.44 g), as can be seen in Figure 1. The Argentinian soybean meal, obtained as a co-product of the oil extraction, is a stable commodity in the feed market, with a global demand which is assumed to remain constant despite the increase in demand for biodiesel in Spain. Hence, it is not involved in the loop.

In Scenario 2, using UCO to produce domestic biodiesel avoids the need to import crude palm oil from the marginal supplier, which is again assumed to be Malaysia, since palm oil stands out as the least expensive oil in the international market and Malaysia is the world’s largest exporter (MPOB 2012). Similarly to Scenario 1, it is assumed that this country reduces palm oil production, and less palm kernel meal is thus available. The gap left in the market supply is filled with meal from Brazil, with the subsequent LUC effects. The resulting loop causes a contraction of palm oil production in Malaysia (-26.31 g) and an expansion of soybean meal production in Brazil (+3.58 g), as shown in Figure 2.

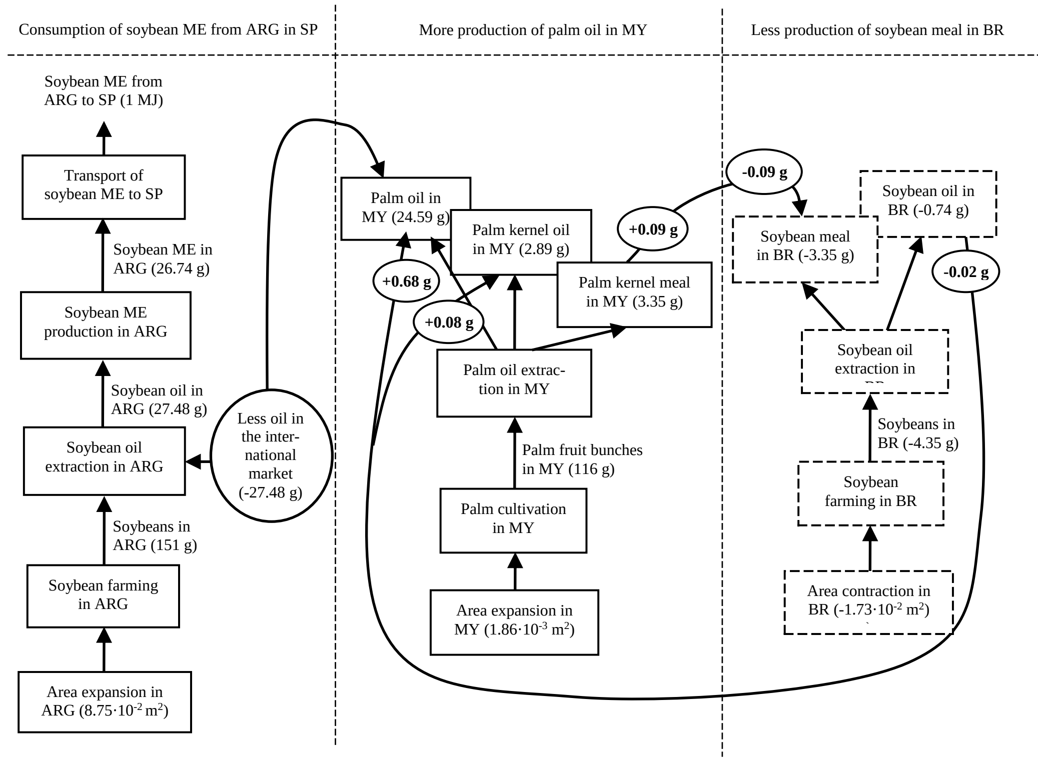

Figure 1. Delimitation of the system under study in Scenario 1. The loop between palm oil and soybean meal in the global market is iterated against zero and the resulting amounts of both are shown in the circles. SP: Spain; ARG: Argentina; BR: Brazil; MY: Malaysia.

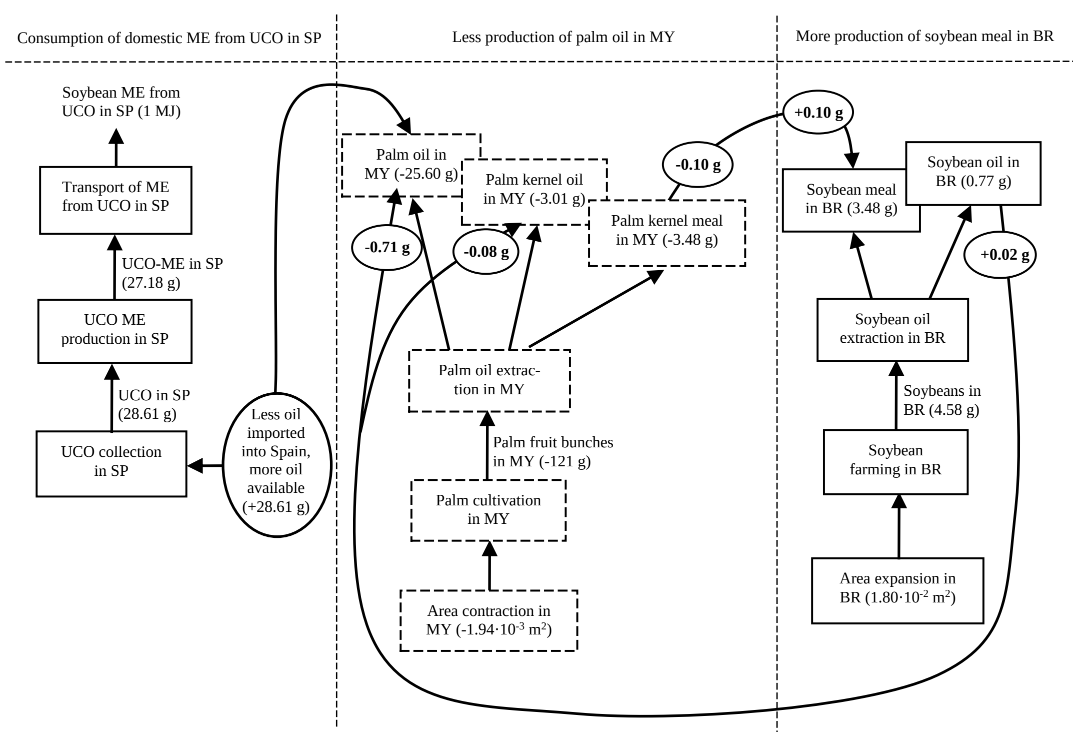

Figure 2. Delimitation of the system under study in Scenario 2. The loop between palm oil and soybean meal in the global market is iterated against zero and the resulting amounts of both are shown in the circles. SP: Spain; BR: Brazil; MY: Malaysia.

2.2. Inventory data

Inventory is drawn from primary and secondary data sources. In Scenario 1, the soybean ME pathway in Argentina is entirely gathered from the inventory of Panichelli et al. (2009). For the export sub-stage, it is considered that the biodiesel is transported first by lorry to the port of Rosario, then to Rotterdam (The Netherlands) by transoceanic tanker, and finally to Valencia (Spain) by lorry. In Scenario 2, all the data relative to the UCO-ME pathway in Spain is provided by the company Bionorte (Asturias, Spain), a representative company of the sector. The distance for the transport of UCO-ME within Spain is that between Bionorte and Valencia, whereas the UCO collection distance is estimated to be 100 km, the same as in the study by Vinyes et al. (2013) analyzing the system consisting of urban collection points. The processes for the production of palm oil in Malaysia and soybean meal in Brazil, common to both scenarios, are taken from the Ecoinvent v.2.2 database (Hischier et al. 2010). The production and provision of all the inputs, including fuel and energy, is included in each sub-stage and taken from the same database, as well as the average transportation systems.

The CML 2 impact assessment method (Guinée et al. 2002) is applied to analyze the impact category Global Warming (GW, 100 years). GHG emissions from dLUC and iLUC are calculated and taken into account for the GW assessment. With this aim in mind, the area diverted to arable land must be multiplied by different emission factors depending on its previous use (forest, grassland, shrub land or other crops). These area values associated with each land transformation are taken from Ecoinvent v2.2 for the palm oil production in Malaysia. Updated values from Prudêncio da Silva et al. (2010) are used for the soybean production in Brazil, since they represent the situation in the Mato Grosso, a state which accounts for 87% of the soybean area in the country. Area values from Panichelli et al. (2009) are considered for the soybean crop in Argentina. Emission factors for each land conversion in each country are calculated by following the guidelines of the IPCC (2006), for a baseline of 20 years. Carbon losses are the result of differences in the carbon content in biomass, soil and dead organic matter before and after the LUC. When deforestation takes place, biomass burning is included, following the process in Ecoinvent v2.2.

3. Results

The GW results show that, without considering the LUC effects, imported soybean biodiesel leads to lower GHG emissions. The contribution of each sub-stage to the impact is shown in Figure 3, depending on whether they cause net GW *input* or GW *output*. Overall, Scenario 1 causes a negative impact (carbon uptake), whereas Scenario 2 causes net CO_2-eq. emissions. As can be seen, this is due to the photosynthesis of the soybean plants and palm trees in Argentina and Malaysia, respectively, which makes GW *input* larger than GW *output* in Scenario 1. In Scenario 2, the only process causing carbon uptake is the soybean farming in Brazil, which is not enough to offset net CO_2-eq. from the decreased palm oil production in Malaysia (through plant photosynthesis) and emissions from the other processes in the system. Carbon uptake by Malaysian palm trees counts as negative GW *input* (implying net emissions) in Figure 3 because 121 g fewer palm fruit bunches are produced due to interactions in the global market. As a result, GHG emissions from producing UCO-ME in Spain are 139% higher, relative to the reference situation of importing soybean ME from Argentina.

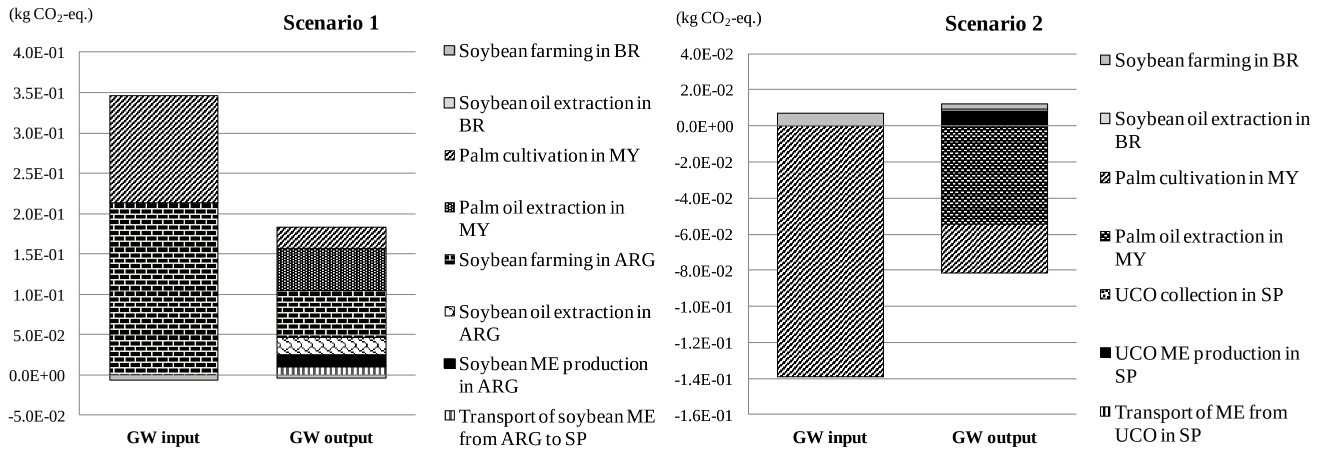

Figure 3. GW results of Scenario 1 and Scenario 2 without considering emissions associated with LUC. SP: Spain; ARG: Argentina; BR: Brazil; MY: Malaysia.

Figure 4 shows the GW results but includes GHG emissions from LUC in their calculation. The UCO-ME pathway then leads to an impact reduction of 103%. In Scenario 1, dLUC in Argentina releases 1137.5 g of CO_2 into the atmosphere, while iLUC in Malaysia is responsible for 221.1 g more, entirely caused by the deforestation of tropical forests; iLUC in Brazil generates an uptake (GW *input*) of 133.8 g of CO_2, because soybean production is falling. Emissions from the entire import chain of soybean biodiesel into Spain are 1028.6 g CO_2-eq., while compensating for the drop in the oil available in the international market with Malaysian palm oil releases another 35.1 g of CO_2-eq. into the atmosphere, even including net carbon uptake in Brazil. As a result, GW *output* is greater than GW *input*, and Scenario 1 generates 1063.7 g of CO_2-eq. On the contrary, Scenario 2 causes a negative overall GW (GW *input* > GW *output*). UCO-ME avoids the production of 28.6 g of CO_2-eq., mainly as a consequence of the area contraction in Malaysia, which causes an uptake of 230.6 g of CO_2, while increasing the agricultural land diverted to soybean in Brazil generates 139.4 g of CO_2.

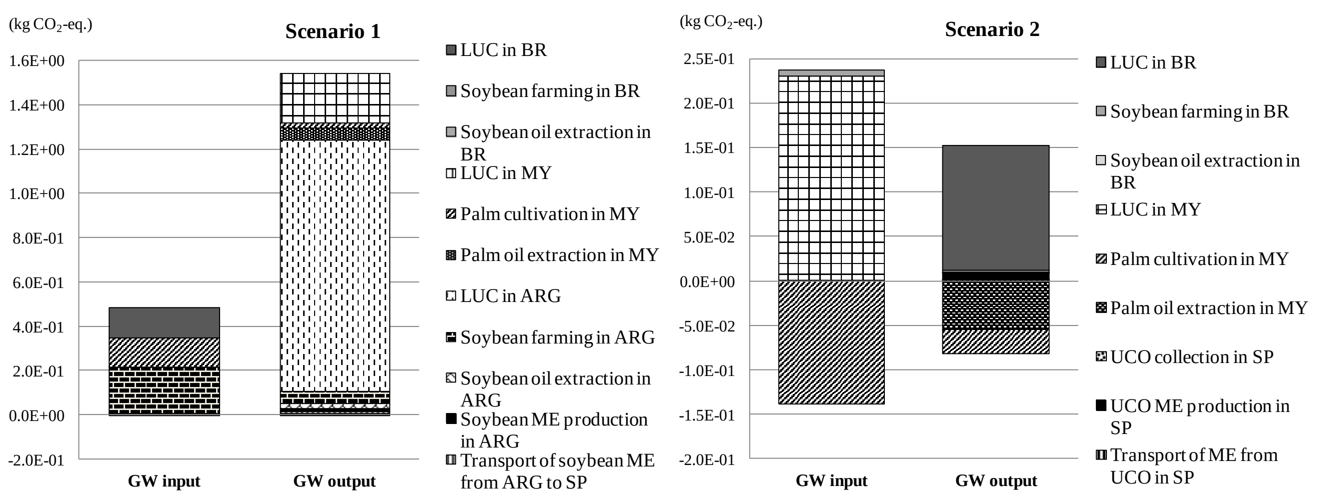

Figure 4. GW results of Scenario 1 and Scenario 2 by taking emissions associated with global LUC into account. SP: Spain; ARG: Argentina; BR: Brazil; MY: Malaysia.

4. Discussion

In Scenario 1, increasing the consumption of biodiesel in Spain by 1 MJ triggers market-mediated responses which, in turn, cause direct land transformation in Argentina, deforestation in Malaysia and area contraction in Brazil. As a result, $8.75 \cdot 10^{-2}$ m^2 of land are diverted to soybean in Argentina, which come from other crops (32%), forest (22%), pasture (27%) and shrub land (19%), according to Panichelli et al. (2009). Similarly,

$1.86 \cdot 10^{-3}$ m^2 of forest are diverted to palm in Malaysia, whereas in Brazil there is an area contraction of $1.73 \cdot 10^{-2}$ m^2. These values are reported in Table 1. Scenario 2 causes an increase of $1.80 \cdot 10^{-2}$ m^2 in the agricultural land in Brazil, while there is a contraction in Malaysia of $1.94 \cdot 10^{-3}$ m^2, as shown in Table 2. Emissions from iLUC are responsible for 92.8% of the GW *input* in Scenario 2, whereas iLUC emissions account for only 8.2% of the overall CO_2-eq. released into the atmosphere by the system in Scenario 1. The most influential sub-stage in Scenario 2 is thus related to an indirect function.

Table 1. LUC results per MJ of biodiesel consumed in Spain in Scenario 1, including direct (in Argentina) and indirect effects (in Malaysia and Brazil).

	Soybean farming in ARG	Palm cultivation in MY	Soybean farming in BR
Transformation from arable land, non-irrigated crops [m2/MJ]	$2.80 \cdot 10^{-2}$	0	$-1.65 \cdot 10^{-2}$
Transformation from forest, clear cutting [m2/MJ]	$1.93 \cdot 10^{-2}$	$1.86 \cdot 10^{-3}$	$-1.73 \cdot 10^{-4}$
Transformation from pasture and meadow [m2/MJ]	$2.36 \cdot 10^{-2}$	0	0
Transformation from shrub land [m2/MJ]	$1.66 \cdot 10^{-2}$	0	$-5.88 \cdot 10^{-4}$
Land Use Change, transformation to arable land [m2/MJ]	$8.75 \cdot 10^{-2}$	$1.86 \cdot 10^{-3}$	$-1.73 \cdot 10^{-2}$

Table 2. LUC results per MJ of biodiesel consumed in Spain in Scenario 2, in terms of only indirect effects (in Malaysia and Brazil).

	Palm cultivation in MY	Soybean farming in BR
Transformation from arable land, non-irrigated crops [m2/MJ]	0	$1.72 \cdot 10^{-2}$
Transformation from forest, clear cutting [m2/MJ]	$-1.94 \cdot 10^{-3}$	$1.80 \cdot 10^{-4}$
Transformation from pasture and meadow [m2/MJ]	0	0
Transformation from shrub land [m2/MJ]	0	$6.12 \cdot 10^{-4}$
Land Use Change, transformation to arable land [m2/MJ]	$-1.94 \cdot 10^{-3}$	$1.80 \cdot 10^{-2}$

These land transformation values for Malaysia correspond to the estimations of Jungbluth et al. (2007), based on historical data for different bioenergy crops. From this, it follows that 100% of the LUC in Malaysia takes place at the expense of tropical rainforests. This type of land conversion leads to substantial carbon losses according to the IPCC (2006), mainly due to the changes in the carbon stock in aboveground and belowground biomass, including dead organic matter, such as wood debris, whose presence is remarkable in areas of tropical rainforests. On the contrary, the establishment of palm oil plantations can generate a gain in the soil carbon stock if it implies reduced tillage with low input, which can even improve the characteristics of the prevailing acidic soils. As can be seen in Figure 4, GHG emissions from iLUC in Malaysia make a significant contribution in both scenarios, since they are enough in themselves to offset the carbon fixation by palm trees during the photosynthesis after the land conversion. However, it must be taken into account that the land transformation values associated with each bioenergy crop are subject to change depending on the temporal and spatial framework considered. As an example, Germer and Sauerborn (2008) found that establishing palm oil plantations in anthropogenic grasslands, which are readily available in Southeastern Asia (long ago converted from forest), can significantly improve the GHG balance of palm oil, after rehabilitation.

Similarly, in our case study, LUC in Brazil occurs at the expense of other arable crops in almost the entire area, but also at the expense of forest (1%) and shrub land (3.4%), according to the situation described by Prudêncio da Silva et al. (2010) for the Mato Grosso. These values are based on historical data for the period 2005-2008. However, soybean expansion has also taken place in Southern States, such as Paraná or Rio Grande do Sul, with a long-standing tradition of agriculture. According to estimations by the same authors, if this second possibility was considered, there would be neither transformation from rainforest nor from savanna. This would even improve the GHG balance of Scenario 2, since the GW *output* would decrease, whereas the GW *input* would be lower in Scenario 1.

As regards LUC in Argentina, land transformation values assumed by Panichelli et al. (2009) are based on 2000-2005 data (Gasparri et al. 2008), a period of great expansion in soybean production in Argentina due to the consolidation of the biodiesel sector in regions such as the EU. It can be assumed though that this situation barely changed during the 2005-2008 period (consistent with data for Brazil). In fact, deforestation still takes place in the region of El Chaco (Aide et al. 2013; Gasparri and Grau 2009). However, considering LUC patterns in the central region of the country (Buenos Aires, Córdoba and Santa Fe) would lead to different results, since there are no native forests and soybean expansion mainly occurs at the expense of other crops.

Considering LUC in Indonesia instead of in Malaysia may have led to different transformation values in Tables 1 and 2 as well, although the emission factors calculated from the IPCC (2006) guidelines for tropical forests should be the same. According to data from the UN-RED programme launched by the United Nations and with the support of the Food and Agriculture Organization (FAO) –among other partners–, in the 20 years from 1983 to 2003, there was a reduction of about 4.9 Mha of forest cover in Malaysia, or an average of 250,000 ha of forest lost annually. However, a recent study by Margono et al. (2014) reveals a loss of primary forest in Indonesia of over 6.02 Mha from 2000 to 2012. Only in 2012, Indonesia lost 840,000 ha of its primary forest, becoming the world's third largest producer of GHG behind China and the US, with 85% of its emissions coming from forest destruction and degradation. This shows that deforestation rates can be highly variable depending on the period considered, and highlights the need to update conversion rates (in terms of ha/year) for each country in databases such as Ecoinvent, in particular for tropical regions in which the palm industry is still expanding. Thus, new rates should be consistent with spatially and temporally explicit observations, since they ultimately determine overall LUC.

This discussion may also contribute to the debate on land use transitions. This concept simply refers to any change in land use systems from one state to another one, while trying to understand the ecological, social and economic circumstances which trigger specific transitions observed in different regions. Specifically, forest transitions have been widely studied, firstly in Europe and the US, and more recently in tropical and sub-tropical regions, in order to explain alternating periods of deforestation and reforestation. According to Lambin and Meyfroidt (2010), LUC is a non-linear phenomenon, which is associated with other biophysical and societal changes and must thus take into account both socio-ecological and socio-economic considerations. For the present study, it has been assumed that LUC effects occur on a global scale, implying that reforestation in one region must be compensated by deforestation in another region, in order to produce the commodities which are no longer produced in the first region. The globalization of timber and agricultural markets is also mentioned by Lambin and Meyfroidt (2010) as an example of a socio-economic force influencing land use transitions. However, due to its dynamic nature, a transition is not a fixed pattern and other political, technological or demographic scenarios could be considered, also including micro-economic considerations about the land tenure system in each region. For instance, it still remains to be seen what will happen with that degraded grassland declared available by Germer and Sauerborn (2008) in Southeastern Asia, whether it will be converted to palm plantations or to tropical forest, depending on government initiatives.

5. Conclusion

As could be expected, biodiesel produced in Spain from domestic UCO performs better than imported soybean biodiesel from Argentina in terms of GW. However, these environmental benefits are not detected if emissions from LUC are not taken into account for the LCA, which can be used as biased results in favor of first generation biofuels, arguing that feedstock production causes carbon uptake due to the photosynthesis and soil carbon storage. Fortunately, most of the recent biofuel policies have mainstreamed iLUC concerns, arising from the increase in the worldwide production of bioenergy. Addressing iLUC effects is thus not temporary, and the consequential approach appears to be an appropriate way of quantifying them within the LCA framework, although there is no consensus on the methodology to apply. The present study shows a case study by using a methodology already developed and accepted within the LCA community, which can contribute to the promotion of the use of biofuels from non-edible biomass (as required by COM 595). This procedure provides additional insights in the analysis of ILUC effects due to a biofuel mandate at country level, which can be used for policy analysis. The results underline the importance of considering indirect effects of biofuel pathways; for this purpose, an analysis of interactions between co-products in the international market is required. A thorough analysis of the uncertainty is recommended in order to reinforce confidence in the comparative assessment, including the variability in land transformation values.

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Environmental impacts of extensive outdoor pig production systems in Corsica

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ABSTRACT

The extensive outdoor pig production systems in Corsica are quite different from the conventional production of Europe. Because of the longer lifetime of pigs and the lower technical performance of animals, the environmental impacts of a kilogram of pig until it leaves the farm gates could be expected to be greater. The diets of pig have to be considered because the animals are fed partly with natural feed like acorns and chestnuts. The hypothesis is made that those feedstuffs which grow naturally without any human mediation do not have any environmental impact. Three Corsican systems with an increasing amount of natural feed in the diet were assessed using Life Cycle Assessment (LCA). The results per kg of pig indicate a decrease of 62-81% for selected impacts from the system with the least amount of natural feeding to the more extensive system. By comparison, the conventional production systems of Europe previously assessed by Dourmad et al. (2012) have intermediate results.

Keywords: outdoor pig production, LCA, natural feed

1. Introduction

The extensive outdoor pig production systems in Corsica are quite different from the standard production systems of Europe which have been described and assessed by Dourmad et al. (2012). The animals are raised outdoors, on extensive areas and partially fed with chestnuts and acorns that have fallen directly from trees. Animals can also be left free in the mountain for a transhumance period. Compared to standard production, the technical performance of the pigs is lower, and their lifetime (18-24 months) longer. Therefore, the Corsican pigs are expected to have a higher environmental impact per kilogram of pig, despite an improved quality (Edward 2005).Their natural feed could partially reduce the environmental impact. It is not cultivated or transported by humans. The chestnuts and acorns grow naturally on trees, with no maintenance.

This study performed an environmental assessment by Life Cycle Assessment (LCA) of three outdoor pig production systems in Corsica. The objectives were to quantify the environmental gain on the kilogram of pig produced, reached by the use of natural feed, and also to identify the specific data to collect on such systems to be able to make an evaluation (Webb et al. 2014).

2. Methods

2.1. Description of the three Corsican pig systems

Three extensive outdoor pig production systems in Corsica were investigated (Table 1) and compared to the average conventional system in Europe (Dourmad et al. 2012). The technical data of husbandry methods was collected from farms. Two systems (Farm1 and Farm2) keep the animals on extensive areas where produced feedstuffs are regularly provided to the pigs by the farmer. The pigs supplement their diet by taking acorns and chestnuts from the ground. In a third system (Farm3) a part of the fattening pigs (64%) goes into the mountains for a transhumance period of 135 days per year. During this period, the pigs are not fed at all by the farmer and are dependent on natural feed found in the mountains.

Animal products sold by the Corsican farms are diverse: fattened pigs, piglets and maiden sows. The systems were compared on their common product with results expressed for fattened pigs and sows.

The systems differed by the type and the proportion of produced feedstuffs in the pig diets: grains from crops cultivated in Corsica (maize, barley …), feeds produced in Corsica with Corsican feedstuffs, and feeds coming from metropolitan France. Their amounts and characteristics were considered to estimate the ingestion by the pigs of purchased feeds (Table 2). The Corsican feeds given to the pigs are specific to extensive diets. It is elaborated to complement the seasonal starchy diets with chestnuts and acorns. It takes into consideration the growing characteristics of the pigs and their energy needs. For this reason, the crude protein content of the feeds is higher than in standard production (average content of 13.4% given by Dourmad et al. 2012).

Per sow and compared to Farm1, the amount of produced feedstuffs given to the pigs is respectively lower by half for Farm3 and higher by 18% for Farm 2 (Figure 1). The main difference between Farm1 and Farm2 is the type of diet and its protein content. Farm1 gives mainly concentrated feeds to sows with less quantity of feed and more protein content than in Farm2. Farm 2 gives grains (barley, maize) which correspond to higher quantities of feed with less protein content. For the three systems, the amount of feedstuffs and protein is largely below the amount given in the conventional system of Dourmad et al. (2012) (1330 kg of feed /sow/year and 28 kg N/sow/year). This could be partly explained by the fact that the pigs also ingested natural feed (chestnuts and acorns) which was not considered. But the total protein amount ingested by sows (even with the natural feed included) is known to be less than in standard conditions.

Table 1. Description of the three Corsican pig production systems assessed and comparison with the average conventional system of Europe described by Dourmad et al. (2012)

Units	Description data	Farm 1	Farm 2	Farm 3	*Conventional system of Europe (Dourmad et al. 2012)*
Pig unit	Products				
	- **Finished pigs (nb/year – *kg LW*)**	**104 – *120 kg***	**65 – *140 kg***	**204 – *120 kg***	***4910 - 113 kg***
	- Primary sows (nb/year – *LW kg*)	10 – *65 kg*	15 – *65 kg*	16 – *85 kg*	
	- Culled sows (nb/year – *LW kg)*	5 – *155 kg*	5 – *170 kg*	16 – *170 kg*	
	- Piglets (nb/year – *LW kg*)	70 – *8 kg*	338 – *20 kg*		
	- Boar (nb/year – *LW kg*)	1 – *176 kg*		0.5 – *176 kg*	
	Pig areas (ha)	170	219	220	*0*
Farrowing unit	Number of sows (nb)	21	30	40	*395*
	Housing	Outdoor / sheds	Outdoor / sheds	Outdoor / sheds	*Building - slatted floor*
	Piglets per sow per year (nb)	5.7	14.1	5.8	
	Produced feed per sow* (kg/year)	546	645	278	*1330*
	Produced feed intake (%):				
	- Growing diet	66%			
	- Finishing diet	23%		93%	
	- Corsican maize	11%	42%		
	- Corsican triticale		7%		
	- Corsican barley			7%	
	- Corsican diet 80-20		51%		
Fattening unit	Housing	Outdoor	Outdoor	Outdoor + transhumance	*Building - slatted floor*
	Age at slaughter of fattening pigs (d)	540	503-720	549	
	Live weight of fattening pigs at slaughter (kg LW)	120	140	120	*113*
	Produced feed conversion ratio[a] (kg/kg)	4.1	7.3	1.4	*2.5*
	Produced feed intake (kg/year) :				
	- Growing diet	61%	2%	15%	
	- Finishing diet	21%	2%	69%	
	- Corsican maize	18%	30%		
	- Corsican triticale		0.4%		
	- Corsican barley			9%	
	- Birth diet		1%	7%	
	- Corsican diet 80-20		37%		
	- Corsican diet 60-20-20		28%		

[a] without natural feed which corresponds to a part of the diet for the Corsican pigs

For the fattening pigs, from weaning to selling, Farm3 is still the most extensive system with lower quantities of produced feedstuffs and nitrogen given to pigs. Farm1 and Farm2 give a close amount of nitrogen coming from the produced feedstuffs, but with half as many feedstuffs for Farm1. This is due to the type of feedstuffs provided with mainly concentrated feeds in Farm1 and with grains in Farm2. The comparison with the European standard production of Dourmad et al. (2012) (267 kg of feed/pig and 6.8 kg N/pig) shows higher amounts of feedstuffs and nitrogen given to the pigs for Farm1 and Farm2 (Figure 1). This is due to the lifetime of the pigs which is three times longer (18-24 months) than in standard production. The amount of feed could also be explained by lower technical performances and a lower adaptation of the feeds to the physiological needs of the animals. However, Corsican pigs also eat natural feed which is not presented on figure 1. By considering it, the difference with conventional production would be greater. Farm3 has lower amounts of feed and nitrogen per pig compared to conventional system because the proportion of natural feed is greater.

Table 2. Characteristics of the produced feedstuffs given to the pigs in the three Corsican farms

		Corsican growing diet Farm 1	Corsican finishing diet Farm 1	Corsican birth diet Farms 2 & 3	Corsican growing diet Farm 2	Corsican finishing diet Farm 2	Corsican growing diet Farm 3	Corsican finishing diet Farm 3	Corsican diet 80-20	Corsican diet 60-20-20	Corsican barley	Corsican maize	Corsican triticale
Ingredients (kg/ton of feed)	Wheat	+	++	+++	+		+++	+					
	Wheat middlings			+				++					
	Bran				++	++							
	Barley	+++	+++	++	++	+++	+	+++					
	Maize	++		+	++	++	+						
	Pea	++	++		++	+		++					
	Rapeseed meal			+	++		+						
	Soybean meal	+	.	+			++						
	Cane molasses					.	.	.					
	Salt	.	.	.	.	.	.	.					
	Amino acids	.	.	.	.	.	.	.					
	Calcium carbonate	.	.	.	.	.	.	.					
	Phosphate	.	.	.	.	.	.	.					
	Corsican Maize								200	200		1000	
	Corsican Barley								800	600	1000		
	Corsican Triticale									200			1000
	Total	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*	*1000*
Nutritional characteristics	Crude protein (%)	25.5	20.8	27.3	24.6	17.7	28.6	19.2	9.7	9.7	10.2	8.1	9.6
	Phosphorous (%)	0.47	0.45	0.51	0.65	0.52	0.54	0.49	0.32	0.32	0.34	0.26	0.35
	Gross Energy (MJ/kg)	25.0	24.7	25.2	25.2	24.8	25.2	24.6	16.0	16.0	16.0	16.2	15.7
	Digestibility of the feed (%)	84	83	82	77	79	83	81	84	85	82	90	88
	Ash of the feed (% MS)	5.0	5.0	6.5	5.0	5.0	5.0	5.0	2.0	1.9	2.2	1.2	1.9

+++ amount > 300 kg / t of feed ; ++ 150<amount<300 kg ; + 50<amount<150kg ; . amount<50 kg

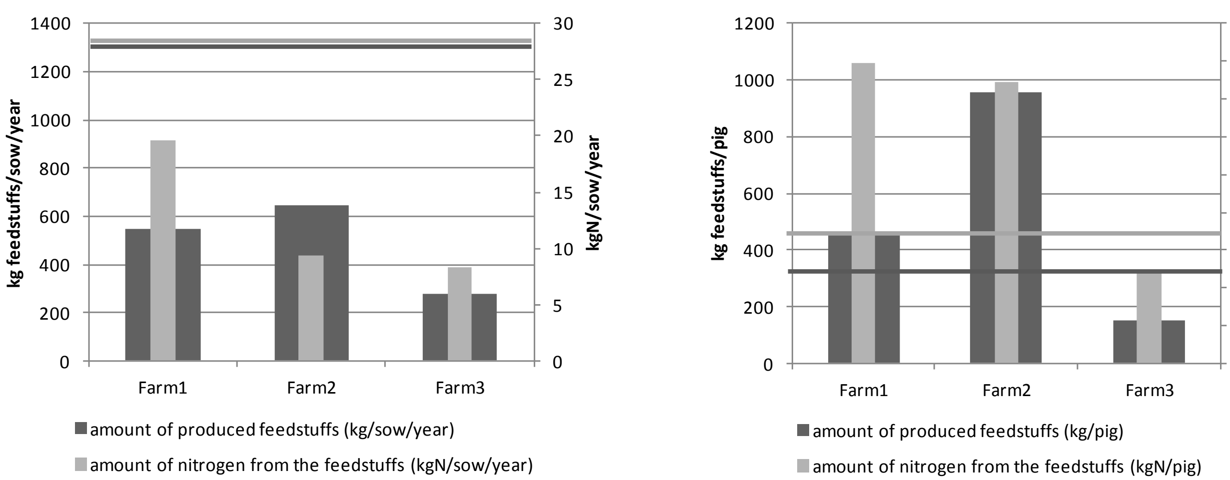

Figure 1. Produced feedstuffs in the diet of sows and fattening pigs of the Corsican systems compared to the standard production of Dourmad et al. (2012)

2.2. LCA assessment

The three extensive outdoor pig production systems in Corsica are assessed, using Life Cycle Assessment (LCA) for the impacts: climate change (kg CO_2eq), acidification (kg SO_2eq), eutrophication (kg PO_4^{3-}eq), energy consumption (MJ) and land occupation ($m^2.year^{-1}$). The environmental impacts are calculated at farm gate, including the production and supply of inputs (feedstuffs, sheds, primary sows, and boars) (Figure 2). The results are expressed per kg of live weight of the two co-products (sows and fattening pigs). At the farrowing unit, an allocation was made between culled sows and piglets and corresponded to the energetic valorization of the feed (Gac et al. 2014): 60% for the culled sows and 40% to the piglets.

The ingestion of natural feeds (chestnuts, acorns) is not considered in the Life cycle perimeter because the fluxes linked to their production are natural (not linked to human activities) and those due to their degradation (NH_3, N_2O and CH_4 gaseous emissions, NO_3- and P losses) would occur without the presence of the pigs.

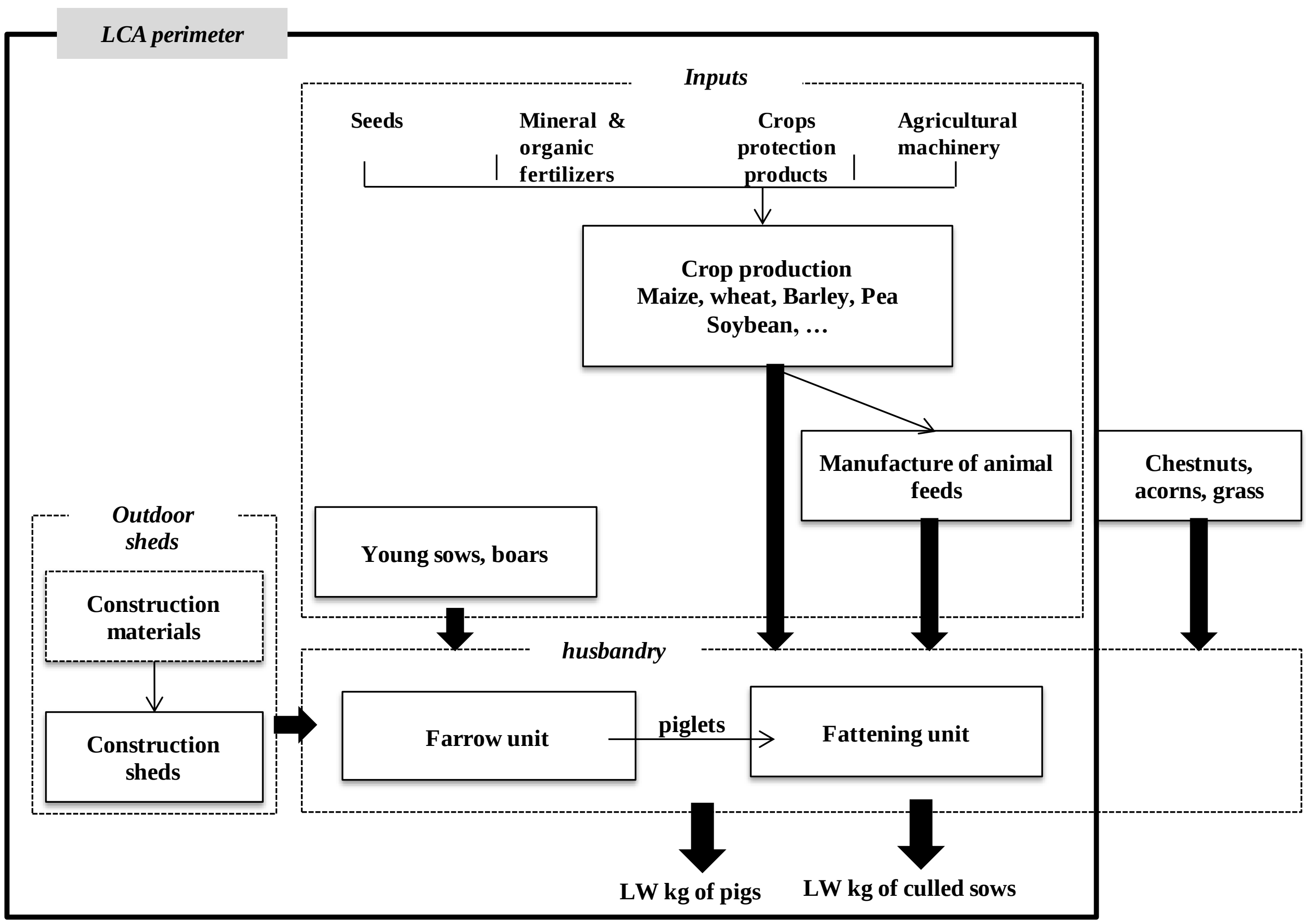

Figure 2. LCA perimeter

Details of the amount and characteristics of the feedstuffs given to the pigs were collected at the farm, and provided by the farmers. The LCA data of metropolitan ingredients of the diet were chosen from a national database (Bouter et al. 2012) and LCA was performed for Corsican barley and maize considering the key data presented in Table 3. For the barley, the straw in Corsica is sold as the grain. An economic allocation was made to consider this specificity: 60% for the grain and 40% for the straw. For the other crops, all the impacts were allocated to the grain.

Table 3. Characteristics of Corsican barley and maize (collected at farm)

Description data	Corsican barley	Corsican maize
N mineral (kg/ha)	40	253
P_2O_5 (triple superphosphate) (kg/ha)	100	139
K_2O (potassium oxide) (kg/ha)	100	
Seed for sowing (kg/ha)	175	26.2
Pesticide (active ingredient) (kg/ha)	0.9	1.23
Diesel (l/ha)	40	91.6
Irrigation water (m^3/ha)	31	2900
Electricity consumption (kWh/ha)		1131
Yield (q/ha)	45	127
Straw (kg/ha)	60	

Sheds were considered for the sows and piglets of the farrowing units with 160 kg of galvanized steel sheets and 0.5 m^3 of concrete per shed. No other infrastructure is used in the three farms. No direct energy consumption is needed for the pig units.

The environmental emissions linked to the excretions of the pigs were assessed with emission factors applied only on the part of the excretion due to purchased feed. The excretion due to natural feed was not considered. The ammonia emissions were calculated with the emission factors of the EMEP/EEA Tier 2 (2009) (eq.1). The method of IPCC Tier 1 (2006) was used to assess, the nitrous oxide emissions (eq. 3.), the methane emissions from enteric fermentation (eq 4.) and the nitrogen leaching (eq. 2.). The IPCC tier 2 (2006) method was used for the methane emissions from the manure management (eq.5). The equation of Nemecek and Kägi (2007) estimated the phosphorous losses.

$$NH_3 = N\ excreted_{\ produced\ feed} \times 0.7 \times EFNH_3 \times 17/14 \qquad eq.1$$
$$NO_3 = N\ excreted_{\ produced\ feed} \times EFNO_3 \times 62/14 \qquad eq.2$$
$$N_2O = (N\ excreted_{\ produced\ feed} \times EFN_2O + NH_3 \times 14/17 \times 0.1\% + NO_3 \times 0.1\%) \times 44/28 \qquad eq.3$$
$$CH_4\ enteric = EFCH_4\ enteric \times AAP \qquad eq.4$$
$$CH_4\ manure = VS_{\ produced\ feed} \times 0.65 \times 0.67 \times EFCH_4\ outdoor \qquad eq.5$$

Where NH_3 emitted = annual ammonia emitted, kg NH_3
NO_3 leached = annual nitrate leached, kg NO_3
N_2O emitted = annual nitrous oxide emitted, kg N_2O
N excreted $_{produced\ feed}$ = annual nitrogen excreted by the pigs coming from produced feed ingested, kg N
0.7 = kg TAN $_{produced\ feed}$ /kg N excreted $_{produced\ feed}$
$EFNH_3$ = emission factor of NH_3, 25%
$EFNO_3$ = emission factor of NO_3^-, 30%
EFN_2O = emission factor of N_2O, 2%
$EFCH_4$ enteric = emission factor of CH_4 enteric, 1.5 kg CH_4/pig /year
$EFCH_4$ outdoor =emission factor of CH_4 from manure, 1%
AAP = Annual Animal Production
VS $_{produced\ feed}$ = volatile solid excreted by pigs coming from produced feed ingested, kg dry matter
0.65 = maximum methane-producing capacity for manure produced by livestock category T, m^3 CH_4 kg^{-1}
0.67 = conversion factor of m^3 CH_4 to kilograms CH_4

The N and P excretions of the pigs were calculated by a mass balance approach considering the nutritional characteristics of the produced feed and the animal performances. The excretion is the difference between the ingestion of produced feedstuffs (the natural feed is not considered) and the body retention of the produced feedstuffs (eq.6 and eq.7 for respectively N and P, Corpen (2003)).

$$N\ Body\ (kg) = e^{(-0.9385 - 0.0145\ Lean\%)} \times (0.915BW^{1.009})^{(0.7364 + 0.0044\ Lean\%)} / 6.25 \qquad eq.6$$
$$P\ Body\ (kg) = 5.3\ BW \qquad eq.7$$

Where BW = body weight, kg

3. Results

3.1. Life cycle inventories

The environmental fluxes are given in Table 4.

The N fluxes (ammonia, nitrous oxide and nitrates) are correlated to the N excretion which is correlated to the N ingestion. Because of this, the relative ranking of the three farms is the same as for the N ingestion (Figure 1).

For the fattening unit, Farm3 which has the most extensive management has N fluxes lower by 86% for ammonia, nitrous oxide and nitrates compared to Farm1. Farm2 has higher nitrogen fluxes of 14% compared to Farm1.

The P fluxes are also reduced for Farm3 compared to Farm1 and Farm2. The methane fluxes are linked to the lifetime of pigs for the enteric component, and to the volatile solid excreted (which is a function of the gross energy intake) for the manure management component. The methane fluxes give less difference between farms than the other environmental fluxes because the methane fluxes are mainly due to enteric emissions (instead of emissions linked to the manure) and the lifetime of the pigs is very close between the three systems.

Table 4. Direct environmental fluxes of the pig units

	Farm1		**Farm2**		**Farm3**	
	Farrowing unit (/sow)	Fattening unit (/pig)	Farrowing unit (/sow)	Fattening unit (/pig)	Farrowing unit (/sow)	Fattening unit (/pig)
N excreted[a] (kg N)	18.4	13.4	6.4	11.5	5.3	2.1
P excreted[a] (kg P)	2.0	1.5	1.3	2.3	0.7	0.1
VS excreted[a] (kg VS)	118.2	98.0	86.2	100.8	79.2	50.3
Ammonia (kg NH_3)	3.6	2.8	1.4	2.5	1.0	0.5
Nitrous oxide (N_2O)	0.7	0.5	0.2	0.4	0.2	0.1
Methane (kg CH_4)	2.2	2.5	2.0	2.7	2.0	2.5
Nitrate (kg NO_3)	22.6	17.7	8.2	15.3	6.5	2.8
P losses (kg P)	0.3	0.3	0.3	0.5	0.3	0.1

[a] part of the excretion linked to the digestion of the produced feedstuffs (without natural feed)

3.2. Life cycle assessment

The LCA results per kilogram of live fattening pig and culled sow are given in Table 5.

Farm3 obtains lower impacts for all the indicators (per culled sow and fattening pig). Compared to Farm1, the result per kilogram of fattening pig are reduced by 64%, 81%, 77%, 62% and 67% for respectively the impacts of climate change, acidification, eutrophication, energy consumption and land occupation.

Farm2 compared to Farm1 has a reduction of impacts of 25%, 24% and 21% for respectively the impacts of climate change, acidification and eutrophication. The result is similar for eutrophication and higher for the land occupation impact.

Table 5. LCA results of the three Corsican farms and comparison with the average conventional system of Europe described by Dourmad et al. (2012)

	Farm1		**Farm2**		**Farm3**		***Conventional system of Europe***
	/kg culled sow	/kg fattening pig	/kg culled sow	/kg fattening pig	/kg culled sow	/kg fattening pig	*/kg fattening pig*
Climate change (kg CO_2eq)	6.9	4.09	6.1	3.03	2.11	1.47	*2.25*
Acidification (g SO_2eq)	87.0	51.5	67.2	38.7	17.0	9.7	*44*
Eutrophication (g PO_4eq)	91.0	53.8	98.9	53.3	25.4	12.3	*19*
Energy consumption (MJ)	35.5	20.2	32.1	15.8	12.4	7.7	*16.2*
Land occupation (m^2year)	11.35	6.43	15.17	7.83	3.78	2.14	*4.13*

For the impacts climate change, acidification and land occupation the respective importance of three main steps in pig production on the LCA result for a kilogram of fattening pig is presented in Figure 3 : the production of the piglets (piglets), the production of the feedstuffs (feedstuffs), the fattening of the pigs (fattening units). The production of piglets on the farrowing unit has a minor incidence on the results. For climate change the step of feedstuff production and the fattening unit have an equivalent impact on the results. This is due to nitrous oxide emissions during the feedstuff production and to both nitrous oxide and methane emissions on the fattening unit. For acidification, the impacts occur mainly on the fattening units with the ammonia emissions while for land occupation it is the feedstuff production which uses land.

For all the differences between systems, both the reduction and the increase occur during the two main steps: production of feedstuffs and the fattening units (Figure 3). It comes from the fact that the feeding strategies impact hugely the excretion from which the environmental fluxes and the impacts are calculated.

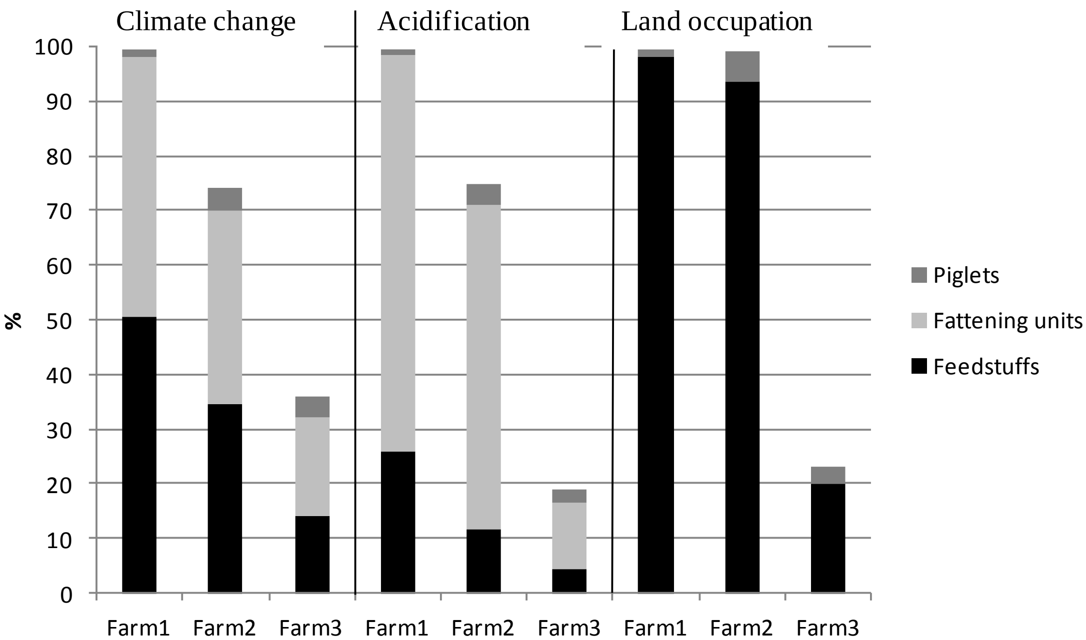

Figure 3. Contribution of the three main production stages to LCA results for a kilogram of fattening pig produced by the Corsican farms

4. Discussion

By comparing the LCA results of the Corsican kilograms of pig to conventional kilogram of pig (Dourmad et al. 2012), Farm3 obtained lower impacts, Farm1 had higher impacts and for Farm2 the results could be higher or lower depending on the impact. As expected, when the impacts for the farms are higher compared to the conventional system, this could be explained by the longer lifetime of the pigs and their lower technical performances. Dourmad et al. (2012) also obtained higher impacts for traditional systems of Europe compared to conventional ones: an increase of respectively 54, 79, 23, 50 and 156% for climate change, acidification, eutrophication, energy use and land occupation. The added value of this study is to have assessed three different extensive systems in which the proportion of natural feed varied. The result indicates that the increase in impact compared to conventional production could be compensated by the use of natural feed in the diet of the pigs.

This is due to the methodological choice of not considering the natural feed in the LCA perimeter. It was determinant in the results. It required being able to specify environmental fluxes without considering grazing and transhumance. The diets were detailed and the associated excretion estimated. The choice of the emission factors applied on the excretion was also a determining factor. Without information on the diets (amounts and composition), global emission factors (Tier1) would have been used, especially for ammonia and methane emissions from manure management. The environmental fluxes would have considered a global average diet without the possibility of extracting natural feeds from the assessment. The specification of the intake and excretion linked to manufactured feeds gave the information that made it possible to conclude on the specific impacts of extensive pig systems in Corsica. The more specific the system, the more detailed the information about the LCI should be, because average references are not valid.

5. Conclusion

This study shows the interest of natural feed in extensive pig systems. By not considering it in the LCA assessment, the environmental impacts of the kilogram of pig at farm gate are reduced and could reach a level below the impacts of conventional production. The results also underline a huge difference among the three systems studied which could change the conclusions of a comparison with conventional production. The sensitivity of the results to the diet of these systems is relevant because the importance of natural feed modifies all inventory data such as the amount of feed, the excretion and the associated environmental fluxes. To reach such a conclusion made it necessary to detail the diet without the natural feed and the associated excretion.

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Environmental improvement of pig production: construction and assessment of eight models of pig farms for the future

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ABSTRACT

In a context where husbandries are questioned on their environmental impacts, optimized systems for the future should be devised to provide the milestones. Eight pig systems were constructed by experts by considering the use of best available techniques (BAT), the modernization of buildings and the improvement of technical performances. The kilogram of pig produced from each optimized system was assessed at farm gate using Life Cycle Assessment (LCA) and the results were compared to an average current pig system in conventional production. The results indicate a reduction of impacts which could reach respectively 39, 43, 26, 26 and 45% for climate change, acidification, eutrophication, land occupation and energy use. The implementation of BATs is limited by their cost. Husbandries should improve their technical performances more than those of the 10% best current husbandries to maintain the current cost of production.

Keywords: future models, pig farms, environmental optimization, LCA

1. Introduction

In a context where husbandries are questioned on their environmental impacts (Petit and van der Werf 2003 ; Krystallis et al. 2009), environmental assessments of pig systems are needed in order to inform makers on the recent situation and to initiate improvements. The results must concern different environmental impacts in order to promote a sustainable evolution by limiting the transfer of pollution. The assessments must also be explained under the agricultural practices which devise possibilities of action for the farmers. Best available techniques (BAT) are formalized and recommended to farmers (Bref 2003). Each one is efficient on at least one major environmental flux and concerns a specific part of the farm (building, manure storage, spreading). Pig systems applying these BATs have been assessed by LCA regarding different feeding strategies (Garcia-Launay et al. 2014) or different manure managements (Prapaspongsa et al. 2010). This study proposes to assess pig systems considering an environmental optimization applied on the whole life cycle of pig production. The purpose is to reach the possible global gain. Because the BATs could not all be applied on one farm (some of them concern the same part of husbandry and can't substitute for each other; costs also represent a limit), priorities must be found in improving pig husbandries. It results in a combination of BATs which could differ from one system to another. This study built eight configurations which resulted in eight models of future pig production systems. Environmental and economic assessments were performed among those systems to evaluate the improvement and its applicability.

2. Methods

2.1. Construction of eight models with an environmental optimization

Eight models of pig systems were constructed for the next 10-15 years by 35 experts with a goal of environmental optimization (Table 1). Experts from administration, research and industry were individually interviewed. They bring complementary skills which are necessary to devise sustainable models of production for the future. The experts took care of different environmental aspects (reduction of impacts on water, air and soil) in the regulation context, but also economic and social aspects (competitiveness, income, quality of life and labor). Their expertises enabled decisions to be made for each system of a combination of BATs.

Table 1. Characteristics of the eight optimized models on the environment

Logics	←---------**Combination of pig and crop production**--------→			←------------------------------**Specialized production**-----------------------------------→				**Outsourcing of farrowing activity**
Models	**1a**	**1b**	**1c**	**2a**	**2b**	**2c**	**2d**	**3**
Size of the pig unit	175 sows – 200 ha	225 sows – 225 ha	250 sows – 120 ha	475 sows – 70 ha	←------------------------1000 sows – 80 ha----------------------→			900 sows – 100 ha
Location in an area with high animal density			Yes		←-----------------------------------Yes ----------------------------→			
Mode of housing	Straw litter for sows and fattening pigs, open building with natural ventilation	←--Closed building, slatted floor and dynamic ventilation--→						
Pig feeding strategy	←--------------Feed production on the farm, use of feedstuffs produced on the farm, use of soy meal not linked to deforestation, substitution of a part of wheat and soy meal by pea ----------------→							Purchase of feeds
Manure management	Composting manure, spreading manure + exportation	Spreading slurry + small biogas plant at the farm with slurry and intermediate crops with energy value	Phase separation by a centrifugal decanter to reduce the excess of phosphorous + spreading the liquid fraction + exportation of the solid fraction	Spreading slurry + exportation	Biological treatment with centrifugal decanter. Spreading of the liquid fraction, exportation of the solid fraction and the sludge	V scraper in the fattening building. Exportation of the solid fraction. Spreading of the liquid fraction	Spreading slurry + participation to a large biogas plant for the excess slurry	Spreading slurry + exportation
Best available techniques applied		←--Bioscrubber--→						
		←---Cover of the slurry pit---→						
			←--Energy efficient equipments--→					
		Use of the heat produced by methanization to heat the buildings		Flare for the storage				

The eight pig models correspond to three logics of production which were identified for future innovative and competitive husbandries by Roguet et al. (2009): the combination of pig and crop production (3 models), the specialized production of pigs (4 models) and the outsourcing of farrowing activity (1 model). Other distinguishing criteria concerned the type of production (7 models with standard pork quality and 1 model 1a with improved quality) and the strategy of manure management in relation to the link of livestock to land (conventional storage and spreading, biogas plant, aerobic treatment). The model with improved quality was chosen to be smaller than the other by its size, with more autonomy for the feeding strategy and the manure management, and with the use of straw for the pigs. It corresponds to a model which is often well received by society as it has an environmental consideration. The level of access to land is determinant for manure management and depends on the agricultural area of the farm but also on its location. For this reason some models defined for territory with a high density of animal production treat their manure for abatement and exportation (1c, 2b, 2c and 2d). For the other models, spreading was considered to be the best way to valorize the manure.

2.2. LCA assessment

The kilogram of live pig at farm gate for the eight models was assessed by Life Cycle Analysis (LCA). The environmental impacts were: Climate change in kg CO_2eq (CC), Eutrophication in kg PO_4^{3-}eq (E), Acidification in kg SO_2eq (A), Energy consumption in MJ (EgC) and Land occupation in m^2year (LO). The LCA scope included the production and supply of inputs, the construction of the building, and the pig breeding (Figure 1). Concerning manure management, the system boundaries integrated the avoidance of the production and application of mineral fertilizer which would be applied on crops if manure were not spread, as described by Nguyen et al. (2010).

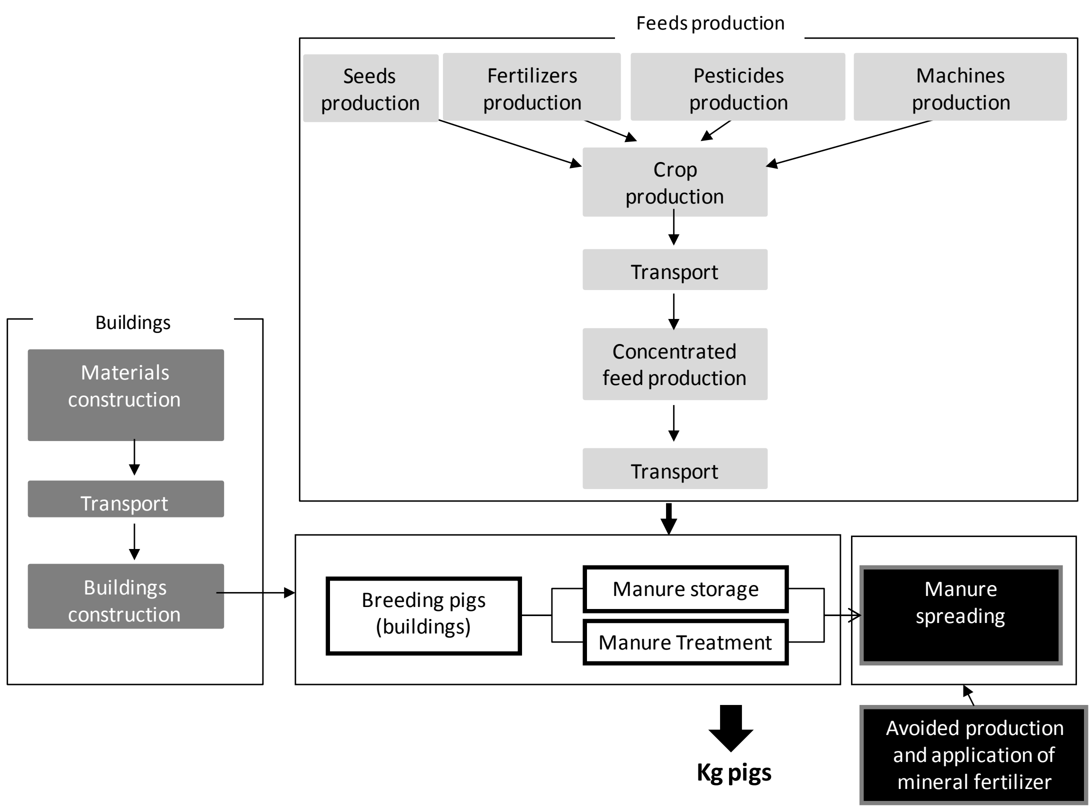

Figure 1. LCA perimeter

In order to measure the reduction of impacts, the optimized models were compared to an average current pig system defined by Espagnol et al. (2012), with the same LCA methodology. The reference system is a fully slatted floor with a classic management of manure (storage under the pigs in the building, external storage in a pit and spreading). Its technical performances are given in Table 5 (husbandries with less than 500 sows). No BAT was considered for this current average system. Its impacts per kilogram of live pig at farm gate are given in Table 4 and correspond to those of a current pig system with less than 500 sows.

Table 2. Models used in the LCI assessment of the optimized systems

Substance emitted / Resource consumed	**Source of emissions / consumer of resource**	**Literature reference for model used**
Ammonia (NH_3)	Animal excretion of nitrogen	CORPEN 2003
	Emissions of NH3 (buildings, storage)	CORPEN 2003
Combustion gas	CO_2	Ecoinvent® v2
Methane (CH_4)	Animal excretion of VS	IPCC 2006 Tier 2
	Emissions from manure management	IPCC 2006 Tier 2
	Emissions from enteric fermentation	IPCC 2006 Tier 1
Dinitrogen oxide (N_2O)	Emissions of N_2O (buildings, storage)	IPCC 2006 Tier 2
Phosphorus	Animal excretion of phosphorous	CORPEN (2003)
	Phosphorous losses	Nemecek (2007)

Table 3. Efficiency of the BATs considered in the optimized systems issued from Bref (2003) and national publication (Guingand et al. 2010)

Best available practices	**Application scale**	**Environmental flux**	**Efficiency at application scale (% abatement)**	**Cross effects**
Bioscrubber	Building	NH_3	50%	Increase of the nitrogen content in the manure
Pit cover	Manure storage	NH_3	70%	Reduction of the manure volume, increase of the nitrogen content in the manure
V scraper	Fattening building	CH_4 N_2O NH_3	100% 49% 40%	Increase of the nitrogen content in the manure
Flare	Storage	CH_4	100%	
Heat pump	Buildings	Energy consumption	65%	
Heat exchanger	Buildings	Energy consumption	30%	
Niches with underfloor heating	Farrowing building	Energy consumption	50%	
Niches for weaned piglets	Buildings	Energy consumption	75%	
Centralized ventilation	Buildings	Energy consumption	60%	
Efficient fan	Buildings	Energy consumption	50%	
Control of the ventilation rate	Buildings	Energy consumption	30%	
Trailing shoes, injector	Spreading	NH_3	Respectively 35% and 70%	
Biogas plant	Treatment	CH_4 and energy	Destruction of 100% CH_4 + production of energy sold and heat used for the building	Energy consumption for the biogas plant. Increase of the nitrogen and phosphorous content in the digestate to spread when there is the use of external inputs (2d)
Improvement of the production of crops	Feeds	Reduction of all the impacts	8.8% on the CC, 3.9% on A, 2.1% on E, 4.6% on EgC and 2.9% on LO	
Reduction of the excretions (N and P)	Excretion	N and P excretions	5% for N and P excretions of systems 1a, 1b and 1c 10% for N and P excretions of the others systems	Reduction of the N emissions of the manure and reduction of the area needed to spread the manure

For the Life Cycle Inventory (LCI) of the optimized models, some improvements were considered comparing them to the current situation. The technical performances of the optimized systems correspond to the 10% of the best French pig husbandries (Table 5). A reduction of nitrogen and phosphorous excretion was applied (Table 3) compared to the current situation considering pretreatment of the feedstuffs and multiphase feeding. The soy meal incorporated in the feeds was considered coming from Brazil but from a location not linked to deforestation (Mosnier et al. 2011). An increased incorporation of pea (10%) into the feeds was chosen instead of a part of wheat (5%) and soy meal (5%). An improvement in crop production was also taken into account based on the result of a prospective analysis by the French agricultural ministry (2011). The efficiencies of the BATs at their application scale are given in Table 3. The models used for LCI are given in Table 2.

For the models with biogas plant (1b and 2d), the emissions linked to the digester and pre and post storage were allocated between the kilogram of pig and the kWh produced considering the energetic content of the inputs in the digester.

3. Results

The results of LCA for optimized system and the baseline scenario are given in Table 4. The optimized models compared to the baseline show reduction of impacts that could achieve up to 39%, 43%, 26%, 26% and 45% for the respective impacts of CC, A, E, OS and EgC.

Table 4. LCA results of optimized pig systems

	LCA results / kg of live pig				
	CML, 2001				Recipe
	Climate change (kg CO_2eq)	Acidification (kg SO_2eq)	Eutrophisation (kg PO_4eq)	Land occupation (m^2.year)	Energy consumption (MJ)
System 1a	2.49	0.047	0.023	4.96	11.57
System 1c	1.54	0.029	0.018	6.49	10.07
System 2a	1.33	0.026	0.017	6.71	9.65
System 2b	1.88	0.027	0.017	4.72	11.97
System 2cb	1.51	0.026	0.017	6.34	10.21
System 3	1.69	0.027	0.018	6.07	11.33
System 1b	1.41	0.026	0.017	6.91	8.95
System 2d	1.31	0.025	0.017	6.21	8.98
Baseline	*2.14*	*0.044*	*0.023*	*6.46*	*16.29*

The reduction of impacts of a kilogram of pig at farm gate is performed mainly during the production of the feed and during the breeding of the pigs.

The improvement of the feeding strategy reduces the impacts among the optimized systems of a range from 18 to 20%, 4 to 6%, 5 to 8%, 2% and 11 to 21% respectively for CC, A, E, LO and EgC. It is due both to the improvement of the technical performances of the pigs and also to the reduction of impacts during the crops production.

The use of BAT in the pig husbandries reduce the impacts for all the optimized systems except 1a from 7 to 19%, 29 to 37%, 12 to 16% and 15 to 23% for respectively the CC, A, E and EgC.

The impacts CC, A and E of the system1a are higher than the other systems despite the fact that BATs are used. This is due mainly to the use of litter in the pig buildings which emitted N_2O (impact on CC) and to the fact that techniques like bioscrubber could not be used in natural ventilation conditions (emissions of NH_3 are not abated and have impact on acidification).

Concerning the spreading of the manure, the difference among the optimized systems and by comparison to the current husbandry shows less difference except for systems 1a, 2b and 2c. The systems 1a and 2b abate nitrogen with respectively composting and biological treatment. The content of nitrogen in the manure to spread is also reduced and changes the comparison with mineral fertilization.

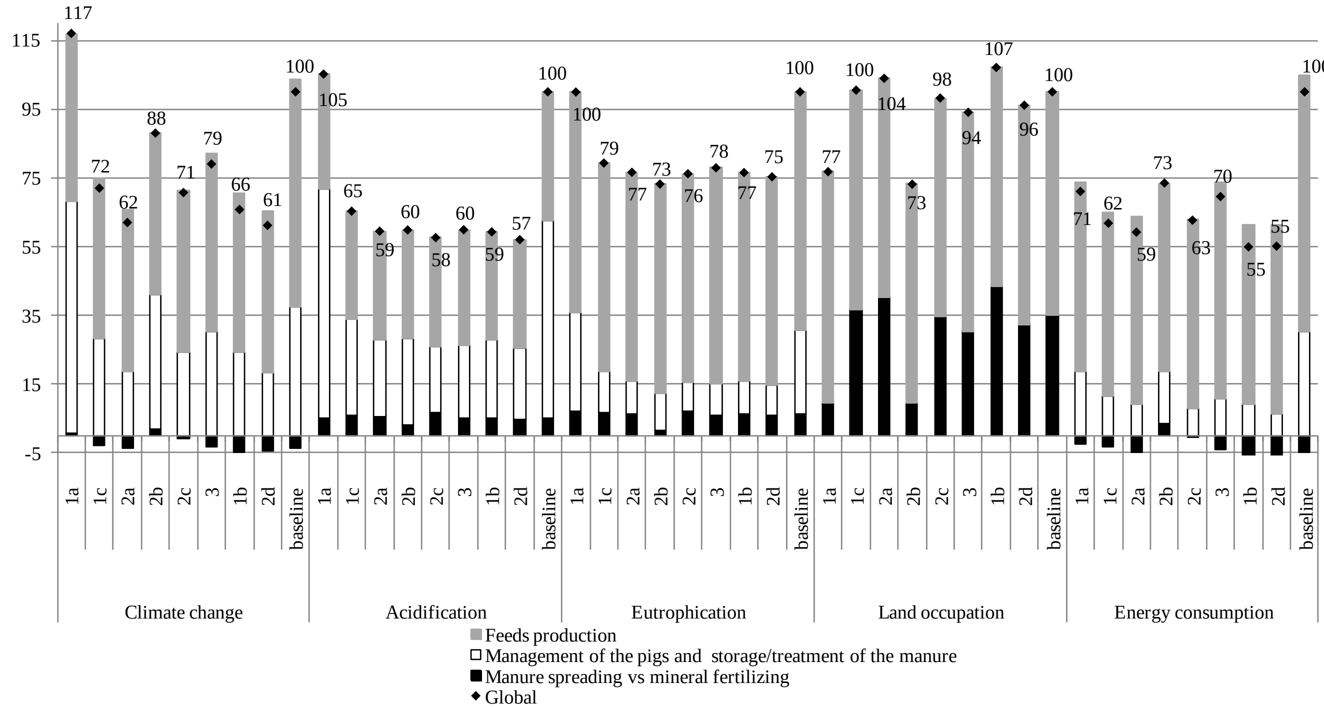

Figure 2. Relative result of LCA in % for the optimized systems compared to the baseline scenario with the importance of three main steps (1/ production and supply of the feeds, 2/ husbandry with the breeding of the pigs and manure management in the building and during the storage/treatment of the manure, 3/ organic fertilization compared to mineral fertilization).

4. Discussion

The result indicates the cumulative reduction of impacts which could be reached by the application of BATs in pig production at farm gate. The next step is to advise the farmers in order to encourage them to go from the current husbandries to the optimized ones. The adoption of BATs will depend on the information they will find. The benefit of BATs at the scale of the pig life cycle is useful and complementary to the efficiency at the application scale because it enables the relative interest of BATs to be measured.

Figure 3 gives the example of three BATs for the reduction of ammonia emissions in pig husbandries. The efficiency at application scale indicates that the pit cover is the most efficient with 70% of abatement versus respectively 50% and 40% with the bioscrubber and the V scraper (Table 3). The reduction rate obtained at life cycle scale shows that the bioscrubber is the BAT which can most reduce the acidification impact. This scale has the advantage of taking into consideration the relative importance of the emissions on which the BAT is applied. It makes it possible to compare different BATs used on different parts of the husbandry (i.e. building, storage …).

Even if the use of BATs indicates possible reduction of the environmental impacts for pig husbandries at LCA scale, the capacity of the current pig units to invest in these is limited because of their costs. The comparison between the three BATs could be completed by analyzing the cost of the reduction of 1 kg SO_2eq/kg of pig (Figure 3). This expression of the results points to the pit cover as being the best in terms of efficiency and the relative cost.

In the future, the cost of BATs could be a limit to their adoption. The bigger units should cope better because they are more likely to be trusted by the banks because of their size and their technical performances. The costs of BATs have been expressed for four types of husbandries: with less than 500 sows, with more than 500 sows, 20% of the best husbandries and 10% of the best husbandries (Table 5).

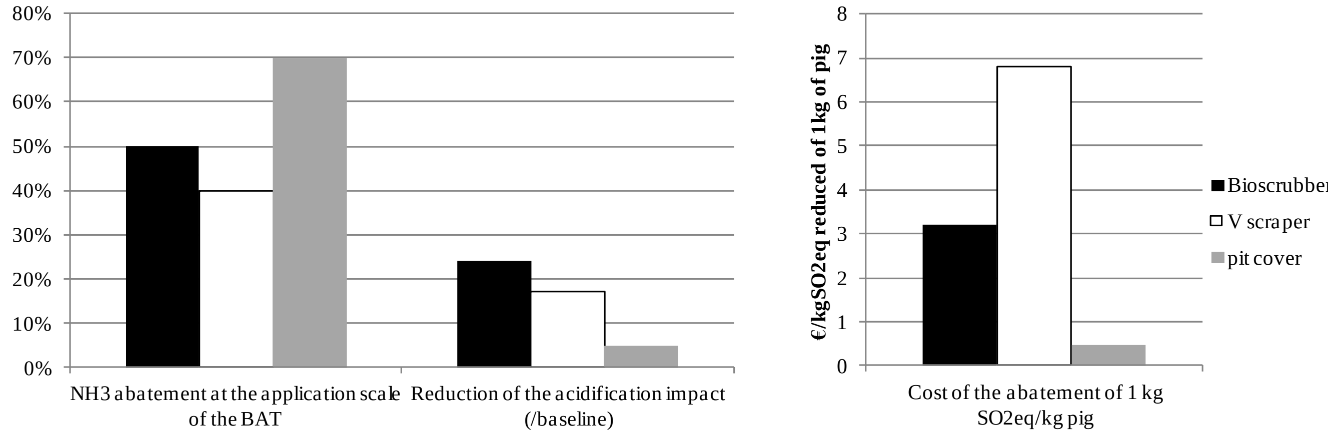

Figure 3. Relative interest for BAT – comparison for bioscrubber, V scrapper and pit cover of the relative interest considering efficiency at application scale (%), efficiency at life cycle scale for acidification (%), the cost of a reduction of 1 kg of SO_2eq per kilogram of pig (€/kg SO_2eq reduced).

For the husbandries with less than 500 sows (representative of current farms with buildings half depreciated), the average cost without BAT is 1.278 €/kg carcass, and goes from 1.359 to 1.438 €/kg carcass with the use of BATs. The BATs without any technical improvement cause an increase in the cost of 0.08 € - 0.15 €. As chosen for the optimized systems of this study, the husbandries of the future might have better technical performances but also increased costs due to necessary investments in modernizing the buildings. With a new building depreciated to 25%, the costs range from 1.300 to 1.453 €/kg carcass depending on the hypothesis made on technical performances. It should be higher than those of the best 10% to access a cost like the current one. This indicates that the implementation of these best practices in existing pig farms is hampered by the additional cost incurred, while the pig price paid to producers is determined in a very liberal and competitive European pig market.

Table 5. Economic assessment of the eight optimized pig systems

	Current pig systems with less than 500 sows*	Current pig systems with more than 500 sows**	20% best husbandries**	10% best husbandries**
Pig produced (/sow/year)	22.34	24.36	25.25	25.94
Weight of fattening pigs (kg)	116.0	116.0	116.7	117.0
Feed conversion ratio (kg /kg)	2.83	2.81	2.73	2.70
Price of fattening pig (€/ton)	184	178	183	183
Working time (h/sow/year)	20	15	15	15
Cost without BAT (€/kg carcass)	1.278	1.314		
Cost with 160€/sow of BAT (€/kg carcass)	1.359	1.384	1.323	1.300
Cost with 315 €/sow of BAT (€/kg carcass)	1.438	1.453	1.389	1.365

*buildings depreciated 50%; ** buildings depreciated 25%

5. Conclusion

This study sheds light on what could be the optimized pig systems for the future by taking into account a reduction of the environmental impacts. BATs have been applied on different parts of the life production of pig including crop production. Important reductions of impacts have been measured and indicate the level of global gain which could be achieved. Different options of BAT combinations could be used and all have results on an impact reduction. This allows the farmers to find the best solution for their system and its location.

The life cycle scale used for the assessment is interesting to measure the relative interest of BATs, and this kind of information is needed by the farmers if they decide to improve their system. The data concerning costs are also critical to the implementation in the field and the study underlines the economic difficulties of applying BATs in current French pig husbandries. The evolution will be correlated by the European pig market on which the price is defined by supply and demand. If all the countries do not decide to invest in BATs, those who do so will be penalized. If the solution is to absorb the additional costs, the improvement of technical performances should be higher than the 10% best current husbandries.

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Connecting the dots: Assessing sustainable nutrition at Nestlé

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ABSTRACT

Sustainable nutrition can be defined as "the physical and economic access to sufficient, safe and nutritious food and water to fulfill dietary and cultural needs to enable an active and healthy lifestyle without compromising the ability of future generations to meet these needs". The combined assessment of the environmental performance (using Life Cycle Assessment) and nutritional performance (using the Nutrient Balance concept) of diets is explored in this paper as a method that can be used to inform strategy setting at Nestlé and to evaluate the progress made in this area. The results of the assessment of diets followed by families living in different countries are presented and discussed.

Keywords: diets, environmental impact, screening LCA, nutrient balance, sustainable nutrition.

1. Introduction

Taking the already established concept of food security as a departing point, the emerging notion of sustainable nutrition can be defined as the "physical and economic access to sufficient, safe and nutritious food and water to fulfill dietary and cultural needs to enable an active and healthy lifestyle without compromising the ability of future generations to meet these needs".

Life cycle assessment (LCA) researchers have already introduced elements of nutritional assessment in the LCA of food products and diets. These include the choice of an adequate functional unit (based on mass or nutritional composition (Kägi et al. 2012); based on protein content (Nijdam et al. 2012)); the comparison of environmental impacts of mainstream diets (omnivore, vegetarian, vegan diets (Baroni 2007; van Dooren et al 2014); or the evaluation of the environmental impacts of changes to healthier diets (Tukker et al. 2011).

Increasingly, research coming from public health and nutrition science is evaluating the link between nutrition and environment. For example, work has been carried out to formulate and recommend healthier, nutritious diets leading to reduced greenhouse gas emissions (Sáez-Almendros 2013; Vieux et al. 2013; Macdiarmid et al. 2011; Wilson et al 2013). The concept of nutrient density can challenge the assumption that excluding dairy products results in diets that meet both nutrient recommendations and result in reduced greenhouse gas emissions (Bruun Werner et al. 2014).

The complexity of food production systems and the potential number of ways of combining food products to achieve the required nutritional quality in a diet pose significant challenges: Is it possible to equate nutritious diets to environmentally sustainable diets (Macdiarmid 2013) (NORDEN 2014)? Do nutrition and environmental sustainability point to convergent or divergent goals (Garnett, 2014)?

This paper contributes to the debate in this field by exploring avenues to combine the nutrient balance and environmental impact assessment of diets. This method can be used by Nestlé at single product, brand or product portfolio perspective; it can also provide a holistic approach to product renovation and innovation.

2. Methods

2.1. Diets evaluated

Menzel and D'Aluisio (2005) inventoried the food supplies purchased in one typical week by 30 families living in 24 different countries. The dietary choices of five of these families were chosen for evaluation using LCA and new tools in nutrition science. It is clear that these are illustrative examples and cannot be taken as statistically reliable samples to represent the typical diets of their respective countries. In this work, as a first approach, it is assumed that all food purchased is consumed over one week in its entirety. Due to the type of data available, it is also assumed that the reported list of food items is not supplemented by additional food items, either already available at home or recently purchased. Table 1 summarizes the total amount of food purchased and total energy content, with the food supplies being further classified into eight categories.

Table 1. Summary of weekly, family food purchases evaluated in this study (Menzel and D'Aluisio 2005)

Labeling	Family A	Family B	Family C	Family D	Family E
Family size	4	4	5	5	4
Family residence	California, USA	North Carolina, USA	Texas, USA	Mexico	India
Total energy content (kcal)	61'306	90'900	110'968	134'694	87'417
Total amount of food purchased (kg)	42.371	78.022	65.814	114.618	51.765
Grains and starchy foods (kg)	6.531	4.522	7.840	15.575	15.358
Dairy products (kg)	4.718	5.383	11.921	14.609	10.388
Meat, fish and eggs (kg)	3.619	6.762	7.644	11.480	-
Fruits and vegetables (kg)	15.176	11.858	12.628	32.648	21.385
Condiments (kg)	0.994	1.925	4.753	2.954	2.156
Snacks and desserts (kg)	1.729	2.604	3.003	1.974	0.196
Prepared food (convenience food) (kg)	3.024	7.987	6.741	1.043	-
Beverages (kg)	6.580	36.981	11.284	34.335	2.282

2.2. Nutrient Balance Concept

The Nutrient Balance Concept (NBC) represents a new approach in nutrient profiling; it goes beyond individual foods and evaluates the nutritional values of multiple foods in meals and total diets. The NBC goes one step ahead of nutrient density by adding the new metric of nutrient balance (QB) to the qualifying (QI) and disqualifying (DI) indices (Fern et al. 2014).

The Qualifying Index (QI) is defined as the ratio of essential nutrients contained in 2000 kcal of a given food relative to the Dietary Reference Intakes (DRI) (USDA 2014) for those nutrients. In the absence of a DRI for a given nutrient, then Adequate Intake (AI) is used instead. The Disqualifying Index (DI) is defined as the ratio of public health sensitive nutrients contained in 2000 kcal of a given food, relative to the Maximal Reference Values (MRV) for those nutrients. The latter nutrients and their respective MRVs are: total fats (<35% of dietary energy); saturated fats (<10% of dietary energy); trans fatty acids (< 1% of dietary energy); cholesterol (<300 mg); and sodium (<2300 mg) (USDA 2010); total sugar (<25% of dietary energy) (Friday and Bowman 2006); alcohol (< 22g) (Foster and Marriott, 2006).

If the QI value is > 1, the food item is considered nutrient dense; if the QI value is < 1, the food item is considered energy dense. If the DI value is > 1, the food item is deemed "compromised" because it contains disqualifying nutrients in values higher than the MRV relative to the energy content of the food item.

Finally, the QB score is calculated as the average proportion of daily values for qualifying nutrients (QI) present in 2000 kcal of a given food. A QB score of 100% is achieved if every qualifying nutrient satisfies 100% of its daily requirement. If the QI for a given nutrient is > 1, then the value is truncated to 1 to avoid bias in the calculations. Table 2 summarizes the calculation of QI, DI, and QB.

The QI and DI values of meals /diets are calculated as the sum of the QI and DI values of the individual component food items, weighted by the contribution of the food items to the overall energy content of the meal /diet.

Table 2. Calculation of QI, QD and QB (Fern et al. 2014)

Equations	Description	Nutrients considered
(1) $QI = \frac{E_d}{E_p} \cdot \frac{\sum_{j=1}^{N_q} \frac{a_{q,j}}{r_{q,j}}}{N_q}$ (2) $DI = \frac{E_d}{E_p} \cdot \frac{\sum_{j=1}^{N_d} \frac{a_{d,j}}{r_{d,j}}}{N_d}$ (3) $QB\,(\%) = 100 \cdot \frac{\sum_{i=1}^{N_q} QI_i}{N_q}$	E_d = daily energy need (kcal) E_p = energy in the qtty. of food analyzed (kcal) $a_{q,j}$ = qtty. of qualifying nutrient (g) $a_{d,j}$ = qtty. of disqualifying nutrient (g) $r_{q,j}$ = DRI of qualifying nutrient (g/day) $r_{d,j}$ = MRV of disqualifying nutrient (g/day) N_q = Number of qualifying nutrients evaluated N_d = Number of disqualifying nutrients evaluated QI = Qualifying index DI = Disqualifying index QB = Qualifying nutrient balance score	**For QI**: folate; niacin; panthotenic acid; riboflavin; thiamin; vitamins A, B6, B12, C, D, E, K; Ca, Cu, F, Fe, Mg, Mn, K, P, Se, Zn; α-linolenic acid; linolenic acid; choline; dietary fiber; protein; water. **For DI**: total fat; saturated fat; cholesterol; trans fatty acids; total sugars; Na; alcohol.

To evaluate the dietary choices of the five example families, the recorded food items (covering the meals for a week) were located in the USDA food composition database (version SR26) (USDA 2013). This database provides the total amounts of the 28 qualifying nutrients and 7 disqualifying nutrients per food item. The DRI data were based on those published by the Institute of Medicine, National Academy of Sciences (USDA 2014). This allowed a comparison of the different results based on the existence of values for all included nutrients. For a more specific assessment, the use of country specific regulated DRI is advised. The MRVs for DI calculation were based on the Dietary Guidelines for Americans (USDA 2010).

2.3. Screening Life Cycle Assessment

The goal of the present screening LCA study was to understand the environmental impacts of real diets of families from varied backgrounds and residing in different locations. The results of this assessment, in combination with those of the nutrient balance concept (see section 2.2) are later used to evaluate the feasibility and implications of such a combined assessment approach.

The functional unit chosen for the assessments was expressed as the “food supplies purchased for consumption at home and to feed one family over the period of one week”. This functional unit was a minimum common characteristic among the 5 diets chosen, acknowledging the fact that the size of the families is not the same, nor the dietary and lifestyle requirements of the various family members. The food supplies consist of raw ingredients which are used to prepare meals at home as well as “convenience” food products which require minimal or no additional preparation.

The scope of the assessment covered the life cycle stages from “cradle to retail”, as shown in Figure 1. The agricultural production of food, the manufacture of ingredients, the manufacture of food products, the retail of the various food supplies and the transportation of materials from one step to the next were considered within the system boundaries. The inclusion of packaging as part of the scope of the assessment would entail taking a number of assumptions derived from the inventory of food supplies (packaging size; packaging materials; level of packaging: primary, secondary and/or tertiary). Packaging was excluded from this screening assessment because the gain in completeness is outweighed by the extensive data collection required and uncertainty introduced. The preparation, consumption and potential wastage of the food supplies along the supply chain and at home were excluded. This is because it is not known how the families combine and use the various food items to prepare meals at home. Accordingly, the end of life of food waste and packaging fall out of the scope of the study.

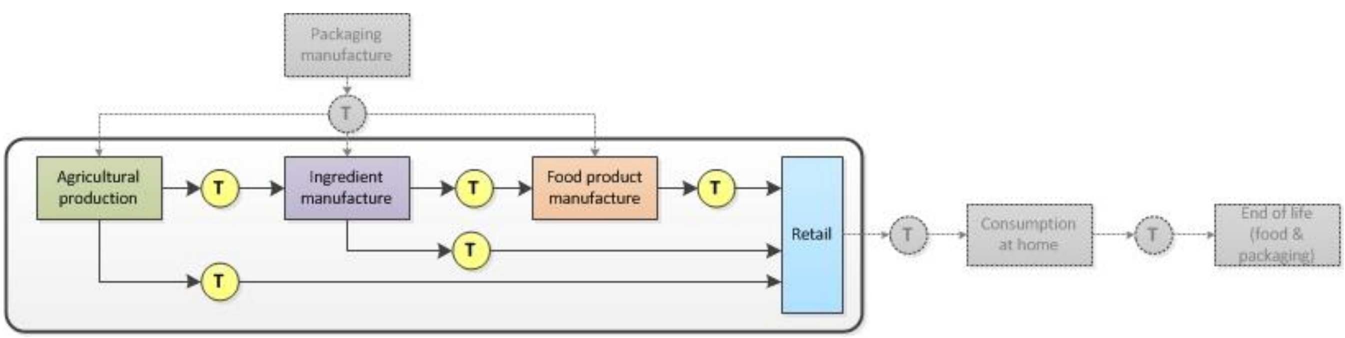

Figure 1. System boundaries considered in the assessment. T = transport.

It was assumed that the transport distance between stages was the same for all 5 case studies (farm to factory, factory to factory, factory to retailer, farm to retailer). Food supplies were considered as regionally produced. In the particular case of food supplies not typically produced in the country, then it was assumed these were imported, and the respective distance and means of transportation were included in the assessment.

Background data (transportation, energy, water supply) and agricultural production data of raw ingredients were taken from the Ecoinvent v.2.2 database (SCLCI 2009). Additional agricultural production data of raw ingredients was taken from Stoessel et al (2012) and the World Food LCA Database Project (Quantis 2014). Recipe composition, food manufacture and retail data were taken from internal Nestlé data repositories.

A screening LCA was performed for each food item reported in the list of purchases of the chosen families. For food items treated as raw ingredients, the scope of the assessment covered the agricultural production and retail; for ingredients and manufactured food items, the scope was extended to include the required intermediate manufacturing stages. Consideration was taken in the supply chain to include ambient or refrigerated transportation, depending on the requirements of each food item. The LCA of each family purchases was then consolidated adding the results of the individual food items.

The screening LCAs were performed using SimaPro 8 (Pré, 2013). Five environmental impacts (at inventory, midpoint and endpoint levels), relevant and adequate to food systems, were evaluated in this study: greenhouse gas emissions (IPCC 2007), freshwater consumption and land use at inventory level, abiotic resources depletion (CML 2014), and impacts on ecosphere / ecosystems quality (Impact 2002+ (Jolliet et al 2003)).

3. Results

3.1. Nutritional balance assessment results

Table 3 shows the relative contributions of the different food groups to the total energy content of the food purchases. It is evident that all families prefer some food groups that contribute the most to the energy content of the food they typically consume.

Table 3. Relative contribution of food groups to the total energy content of the evaluated weekly food purchases

Relative contribution (%)	Family A	Family B	Family C	Family D	Family E
Grains and starchy foods	29	6	18	28	51
Dairy	7	7	9	11	11
Meat, fish and eggs	14	9	12	9	0
Fruits and vegetables	15	15	8	13	22
Condiments	7	2	24	18	14
Snacks and desserts	10	3	10	6	1
Prepared foods	12	10	14	4	0
Beverages	6	48	5	10	1

Table 4 summarizes the results of the NBC analysis of the five family food purchases. The results clearly mirror the different choices the families make. The QB values in the gray fields are later used in the comparison with the environmental impact indicators.

Table 4. Summary of QI, DI, and QB values for the overall weekly, family food purchases and food groups

NBC element	Indices	Family A	Family B	Family C	Family D	Family E
Overall diet	QI	1.4	1.1	1.0	1.0	1.2
	DI	0.9	0.9	1.1	0.9	0.4
	QB	**56**	**40**	**28**	**36**	**68**
Grains and starchy foods	QI	1.4	1.3	1.7	1.0	1.0
	DI	0.5	0.5	0.3	0.3	0.3
	QB	52	52	56	53	40
Dairy	QI	1.3	1.2	1.3	1.1	0.8
	DI	1.3	1.4	1.2	1.3	1.2
	QB	40	40	40	40	28
Meat, fish and eggs	QI	1.1	1.4	1.4	2.4	-
	DI	1.6	1.8	2.1	1.9	-
	QB	40	44	48	52	-
Fruits and vegetables	QI	2.6	2.1	1.5	2.3	2.4
	DI	0.6	0.8	1.0	0.8	0.4
	QB	46	72	68	76	72
Condiments	QI	0.6	0.3	0.2	0.1	0.4
	DI	0.9	1.1	1.2	1.1	1.0
	QB	28	8	4	4	8
Snacks and desserts	QI	0.8	0.5	0.6	0.3	1.0
	DI	0.6	0.9	0.8	0.8	0.5
	QB	32	8	24	4	24
Prepared foods	QI	0.8	0.9	0.9	0.7	-
	DI	1.2	1.2	1.2	1.9	-
	QB	40	32	36	36	-
Beverages	QI	1.2	0.9	1.2	0.3	0.7
	DI	0.6	0.7	0.9	0.7	0.6
	QB	24	12	40	4	24

The overall results indicate that Family C achieved the lowest nutrient balance score (QB = 28) and Family E the highest score (QB = 68). None of the evaluated weekly purchases reached the maximum QB score (100), neither at food group level, nor at the overall level. This means that the recommended nutritional requirements were not met for the amount of energy content of the food purchased. For the various values of QI calculated, there is a range of QB values and vice-versa. The values of QB for the various groups of foods reflect the fact that qualifying nutrients are present in varying quantities. The overall QI values of all families are at or slightly above 1, indicating that the choices between energy-rich and nutrient-rich foods are on average evenly distributed. The DI values are also distributed slightly below or above a value of 1.0 for the DI. A comparatively small value for is observed in the case of Family E. The lacto-vegetarian diet followed by this family contributes to reduced values of fat and cholesterol, which are accounted for in the calculation of DI.

Figure 2 presents the overview of the nutritional qualities of the diets by positioning them in a 2-coordinate map according to their QI and DI values. The value of the calculated nutrient balance QB is also shown in the map. The use of QI and DI allows the mapping of meals/diets in relation to their components. The mapping reveals in which quadrant meals/diets fall, based on their relative quantities of nutrients and energy. Changes in the relative contributions of food items to meals/diets will result in shifts of the QI-DI position and, thereby, reveal the nutritional impact of dietary changes. As a consequence, the NBC tool allows the predictive modeling of dietary choices or choices motivated by non-nutritional needs.

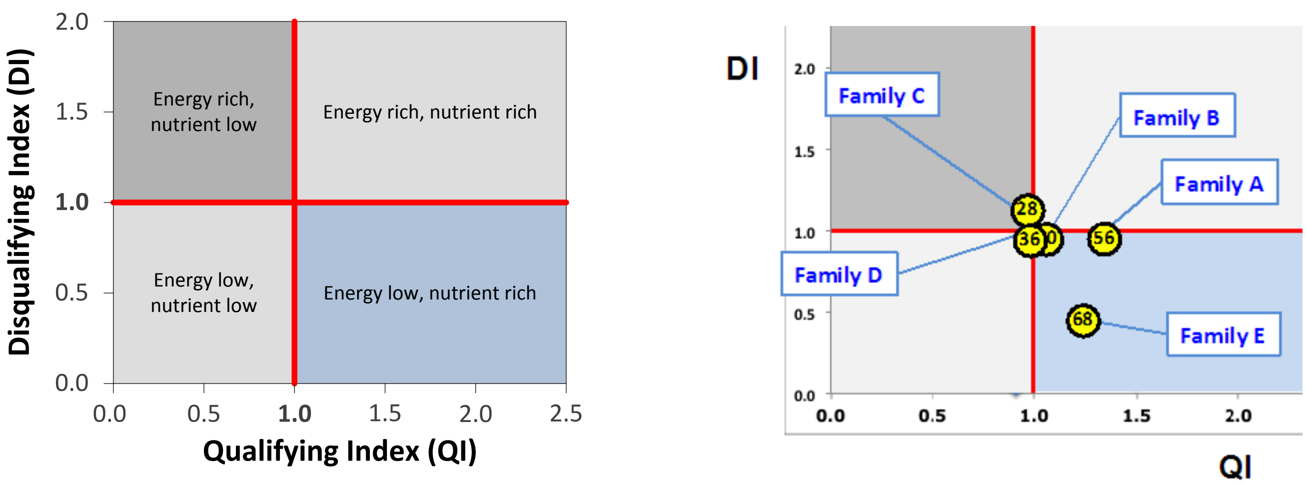

Figure 2. Comparison of the QI-DI positions and nutritional quality of the five weekly, family food purchases. The inserted numbers give the nutrient balance of each diet (QB).

3.2. Environmental impact assessment results

The results of the environmental impact assessment are summarized in Table 5. Figure 3 depicts the relative contribution to the overall environmental impacts of agriculture, transport, manufacture and retail. Figure 4, in turn, shows the relative contribution of the various food groups to the overall environmental impacts.

Greenhouse gas emissions are the lowest in the case of Family E, who follow a lacto-vegetarian diet (dairy products contribute to 31% of total GHG emissions). The high consumption of meat and dairy products for the Families B, C and D, in turn, contribute to around 50% of their GHG emissions.

The results seem to indicate another trend, where large consumption of fruit and vegetables is correlated with large freshwater consumption. This is the case for the Mexican family (Family D), where fruit and vegetables are consumed in large quantities, and account for 71% of the freshwater consumption. On the other hand, 62% of freshwater consumption for the Indian family (Family E) comes from grain consumption. It is necessary to acknowledge that to date water inputs have not been rigorously collected in the preparation of life cycle inventories from different sources. This might introduce a certain bias towards more recent datasets that do include exhaustive information on irrigation, as is the case for fruit and vegetables data used in this assessment (Stoessel et al 2012).

The results for abiotic resource depletion reflect the various levels of energy intensity required at the different stages in the food supply chain. For all cases, farm processes contribute to 70-75% of the results for this impact indicator. It has to be noted that packaging and preparation (which are excluded in the present assessments) can contribute strongly to this indicator, as experienced from previous assessments.

Table 5. Summary of environmental impact results for the weekly, family food purchases

Environmental impact	Family A	Family B	Family C	Family D	Family E
Greenhouse gas emissions [kg CO_2-eq/family*week]	83.65	178.03	136.48	140.61	50.96
Freshwater consumption [m^3/family*week]	3.17	6.10	8.89	10.77	5.37
Abiotic resource depletion [kg Sb-eq/family*week]	0.24	0.50	0.43	0.56	0.20
Land use [ha*year/family*week]	182.25	312.77	271.74	203.69	111.37
Ecosystems quality [PDF*m^2*year/family*week]	7.41	33.11	13.60	31.24	3.30

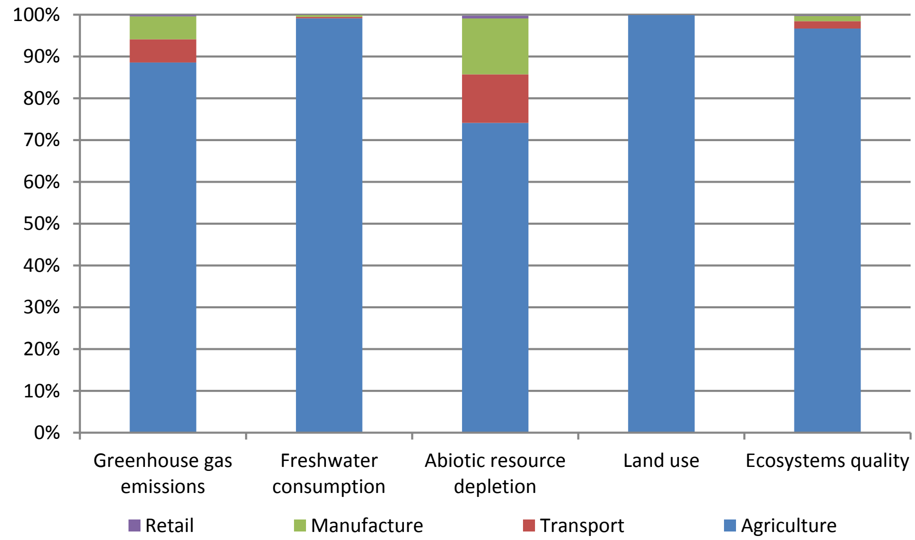

Figure 3. Relative contribution of the life cycle stages to the overall impacts. Average of the 5 assessments.

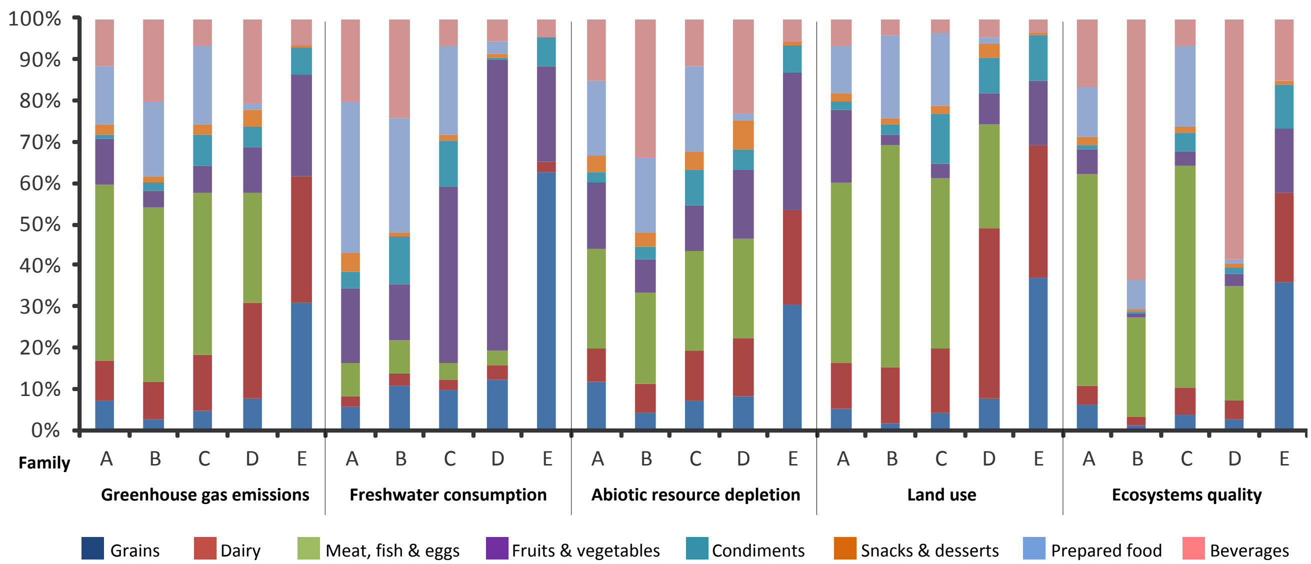

Figure 4. Relative contribution of the food groups to the overall environmental impacts.

Similarly to the trend observed for GHG emissions, land use is driven by the consumption of animal based products. 55% to 67% of the results of this impact are attributable to animal rearing and the cultivation of animal feed for the American families (Families A to C) and Family D. In the case of Family E, given the high proportion of plant-based foods consumed, the cultivation of grains, fruits and vegetables accounts for 53% of this impact. Ecosystems quality is a compound indicator taking into account eutrophication, acidification and ecotoxicity. In all cases, the agricultural activities are the largest contributor to this impact (around 90%).

From Figure 3, it can be seen that the largest contributor to the overall impacts is the agricultural phase. It is necessary to mention that this distribution is influenced by the share of food purchased as raw ingredients (fruit, vegetables, grains and meat) and used to prepare meals at home (23% to 44% of the food supplies). Transport is noticeable in the results for GHG emission and abiotic resource depletion, which is consistent with the use of fossil fuels to power different means of transportation. The contribution of the manufacturing step follows a similar pattern, which is governed by the use of electricity and fuels in the various operations of food processing.

3.3. Combined assessment: environmental impacts and nutrient balance

The results of the combined assessment of environmental impacts and nutrient balance are shown in Figure 5. The environmental impact results are weighted by the QB score of each of the weekly, family food purchases. Figure 5 presents the results normalized with regards to the maximum value for each environmental impact. Families A and E show consistently lower environmental impacts per unit of QB score compared to Families B, C and D. In turn, Families C and D alternatively show the maximum environmental impact per unit of QB score. This reflects not only the differences in dietary choices of the families but also highlights the fact that establishing a direct correlation between reduced environmental impact and high nutritional quality of a diet is not necessarily a straightforward task.

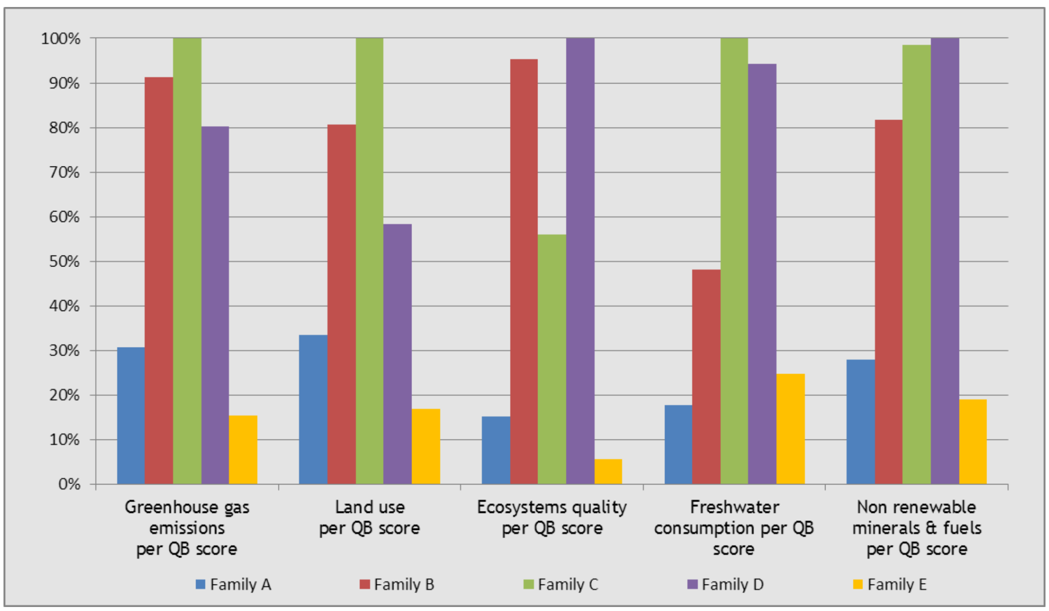

Figure 5. Environmental impact per QB score for each weekly, family food purchases and normalized to the maximum value.

4. Discussion

The presented analysis of the NBC mapping of five family food purchases is a contribution to gain in knowledge on the connection of environmental impact to the nutritional quality of a whole diet. The example of the five families shows that there can be a correlation between the choices leading to a nutritional quality increase and environmental impact reduction. This form of analysis stresses and illustrates the outcome of people's choices.

The USDA food composition database comprises around 8'200 food items covering nearly all food groups. It represents a large "basket" to choose from. However, given that not all the families chosen for the assessment reside in the United States, the food items are not necessarily comparable in all cases, and some were missing from the USDA database. In those cases, the closest match of a food item was substituted for the food item registered in the weekly shopping list. In other cases, the food item was reconstituted according to a matching recipe; then the ingredients where chosen from the USDA database. The "street food" items listed by Family E were not included in the assessment due to a lack of data describing these in more detail; moreover, this group of food items represents one out-of-home dinner at a restaurant (7% of the food expenditure for one week).

The use of a single food composition database and nutritional reference values makes the results comparable. The energy needs of the family members were calculated on the basis of gender, age, weight, heights and physical activity. However, the reference values were taken for an adult. Information for the specific allocation of food products to family members was not available. Moreover, the potential amount of food consumed by any one of the family members could not be taken into account. However, if reference values for families were established and validated, these could be used, resulting in a more representative description of the real situation.

The QB values calculated give an insight into the nutritional quality of food consumed by the families evaluated in this study. For example, Family C shows the strongest mismatch of the essential nutrients balance in their diet. The essential nutrients meet only 28% of the requirements calculated for the amount of energy contained in the food purchased. On the other hand, Family E shows the highest values (QB = 68), which indicates that in this case, the vegetable-rich diet appears to contribute to a more balanced nutrition. It is noteworthy to restate what was mentioned already in Table 2 when explaining the NBC methodology: QI, DI and QB are values calculated as arithmetic means. Therefore, relatively high and low values are likely to be canceled off as part of the calculations and it is not possible to pinpoint, using just the final scores, which nutrients are consumed adequately or not, although this data is generated as part of the model. Complementarity of food components achieve a better coverage of the essential nutrients and will result in a higher QB score.

Figure 4 clearly indicates that the environmental impact of dairy products should not be mixed up or aggregated with those of meat products. A comparison of the environmental indicators for Families A and E shows favorable outcome for Family E compared to Family A. This is due to Family E excluding meat but having a higher amount of dairy products in their diet. A similar conclusion was attained by Bruun Werner et al. (2014) when evaluating the significance of dairy products in realistic dietary choices in Denmark. If the optimization of a diet with regards to reduced GHG emissions implies the exclusion of dairy products, nutritional consequences may ensue. It might not come as a surprise that the improvement of a diet by increasing the energetic contribution of vegetables also leads to an improvement on the environmental indicators.

The results obtained in this study are not necessarily comparable to what is reported in published papers given that the nutrient balance concept is new. Whereas methods for nutritional profiling of diets are more prescriptive, the LCA methodology allows for flexibility depending on the goal and scope of the studies. A preliminary comparison could be done with regards to the GHG emissions associated with food consumption. In this study, GHG emissions range between 1.8 and 6.4 kg CO_2-eq per capita per day. At a national average diet level, reported values for GHG emissions are similar in magnitude: 7.1 kg CO_2-eq per capita per day for Europe (Tukker et al. 2011); 8.4 kg CO_2-eq per capita per day for the United States (Weber et al. 2008); or 4.1 kg CO_2-eq per capita per day for France (Vieux et al. 2013).

5. Conclusion

The FAO (Burlingame and Dernini 2010) sets a clear direction for the task ahead in the area of sustainable nutrition by affirming that sustainable diets are those "with low environmental impacts". The combined environmental impact and nutritional assessment of real diets provides a benchmark and an insightful starting point. This first attempt to assess the nutritional quality of whole meals and diets using the Nutrient Balance Concept has already produced new and unexpected insights. The use of country specific food composition databases and the specific daily reference intake tables of a studied country permit to target the analysis for decision on food production and food consumption taking personal choices of consumers into account.

Further work will focus on expanding the number of diets assessed and the scope of the LCAs (inclusion of packaging, food preparation and food waste); identifying patterns, hot-spots and trade-offs. This will lead to build the evidence base for recommendations and guide product development at Nestlé.

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Considering human exposure to pesticides in food products: Importance of dissipation dynamics

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ABSTRACT

The general public is continuously concerned about effects from pesticide exposure via residues in food crops. However, impacts from pesticide exposure are mostly neglected in food product-related LCAs. Time-to-harvest and dissipation from crops mainly drive residue dynamics with dissipation as the most uncertain aspect in characterization modeling. We analyzed measured half-lives (n=4513) with 95% falling between 0.6 and 29 days. With ~500 pesticides authorized alone in the EU for several hundred crops, however, experimental studies only cover few possible pesticide-crop combinations. Therefore, we estimated dissipation from measured data and provide reference half-lives for 333 pesticides applied at 20°C under field conditions. Our framework allows for detailed explorations of dietary choices in LCA with respect to health impacts from pesticide exposure via crop consumption. The next step is to include pesticide exposure via crop consumption along with improved pesticide dissipation data into existing LCIA methodologies for consideration in future LCA studies.

Keywords: Pesticides, human exposure, crop consumption, dissipation half-lives, life cycle impact assessment

1. Introduction

A main concern of the general public in various regions including Europe and the U.S. is related to long-term or chronic effects from low-level exposures to pesticides (Pretty 2005; Slovic 2010). Human population-level exposure predominantly occurs from residues in food crops, supplemented by exposure from pesticide fractions lost from agricultural fields after application (European Commission 2006; McKinlay et al. 2008). However, in almost all existing food product-related life cycle assessment (LCA) studies human and ecosystem impacts from exposure to pesticides (or other potentially toxic chemicals) are not considered at all or considered negligible (Heller et al. 2013). This is partly due to the perceived high uncertainty associated with impacts from exposure to toxic chemicals (Finnveden et al. 2009). A framework to characterize human health impacts of individual pesticides applied to different agricultural food crop types grown in Europe has been recently published (Fantke et al. 2011, 2012a). This framework is based on a set of interconnected compartments, for which pesticide fate and residues are calculated by solving s flexible set of differential mass balance equations with assumed first order kinetics by means of matrix algebra. In this study, time between pesticide application and crop harvest along with pesticide dissipation half-lives in crops have been identified as key aspects driving residue dynamics, which is consistent with similar assessments focusing on crop residue dynamics (Juraske et al. 2009; Rein et al. 2011). Whereas the influence of time to harvest has been analyzed and parameterized for existing life cycle impact assessment (LCIA) models, pesticide dissipation in food crops remains one of the most uncertain aspects in characterizing human toxicological impacts of pesticides via food crop consumption (Juraske et al. 2008; Fantke et al. 2012b). In the present study, we aim at reducing uncertainty associated with pesticide dissipation in crops, thereby also facilitating a feasible starting point for including human health impacts from pesticide exposure via food crop consumption in future LCA studies.

2. Methods

We start from following the framework proposed by Fantke et al. (2012a) to calculate human health impact scores for pesticides applied to agricultural fields in Europe. This approach is based on estimated agricultural application data for the five most extensively used pesticides per crop and country in 2003 for 24 member states of the EU as of 2004 (EU24) from a collaboration report between Eurostat and the European Crop Protection Association (European Commission 2007). Health impact scores of pesticide i applied to crop c in a specific year, $IS_{i,c}$ (DALY/year), are calculated from the total mass applied in this specific year, $m_{i,c}$ ($kg_{applied}$/year), the pesticide residue fraction in crop harvest, $hF_{i,c}$ ($kg_{residue}/kg_{applied}$), the food processing factor, PF_c ($kg_{intake}/kg_{residue}$) accounting for reduction of residues in crop harvest due to e.g. washing or cooking, the dose-response slope factor, $ß_i$ (incidence risk/kg_{intake}), and the severity factor expressed in disability-adjusted life years as a composite measure for overall population health impacts, SF (DALY/incidence):

$$IS_{i,c} = m_{i,c} \times hF_{i,c} \times PF_c \times \beta_i \times SF \quad \text{Eq. 1}$$

Fantke et al. (2012a) reported an overall model output uncertainty range, i.e. a variance in health impact scores, associated with contributing input variables along the impact pathway (Eq. 1) between 4.75 and 838,505 DALY/year across EU24 in 2003. Main source of uncertainty in the mass balance resulting in residual fractions of pesticides in crop harvest is the first order rate coefficient representing dissipation from crops. Hence, we focus in this study to reduce uncertainty related to dissipation in crops. Uncertainty of any model input and output variable x can be characterized by its squared geometric standard deviation $GSD^2(x)$ and related probability of 95% = $\{x/ GSD^2(x) < x < x \times GSD^2(x)\}$. We use $GSD^2(x_k)$ for input variables $x_k \in \{m_{i,c}, hF_{i,c}, PF_c, ß_i, SF\}$ as provided by Fantke et al. (2012a) to obtain $GSD^2(IS_{i,c})$ of model output (health impact scores $IS_{i,c}$):

$$\mathrm{GSD}^2(IS_{i,c}) = \exp\left(\sqrt{\sum\nolimits_{k=1} (\ln[\mathrm{GSD}^2(x_k)])^2}\right) \quad \text{Eq. 2}$$

The uncertainty related to the calculation of pesticide residues in crop harvest is reported to be $GSD^2(hF_{i,c}) = 27$ and to correspond to a contribution of 29.6% to the output-related uncertainty of $GSD^2(IS_{i,c}) = 412.73$. To reduce $GSD^2(hF_{i,c})$ and thereby also $GSD^2(IS_{i,c})$, we systematically built an inventory database of existing experimental studies providing dissipation half-lives in agricultural food crops and other plants (Fantke and Juraske 2013). Measured dissipation half-lives from 811 scientific studies (>99% peer-reviewed) were available or calculated from concentration-time curves for 346 pesticides applied to 183 crops (n = 4513) with 95% of all half-lives falling between 0.6 and 29 days. However, with almost 500 pesticides authorized alone in the European Union for use on several hundred crops, analyzed experimental studies still cover only a small fraction of possible pesticide-crop combinations.

In response to this need, we developed models to estimate dissipation in crops based on the review from Fantke and Juraske (2013). We first characterized measured pesticide dissipation half-lives by describing their distribution and by determining the influence of temperature from a subset of 1030 data points with reported study condition average air temperatures. We then provided recommended geometric means of dissipation half-lives at 20°C and 95% confidence intervals for 333 reported pesticides, and used multiple imputations for substituting missing temperature data. Next, we proposed a regression-based model to predict dissipation half-lives for pesticides as a function of temperature, substance chemical class, selected substance properties and plant characteristics for all pesticides where reported data were not available. We finally evaluated model prediction performance using sums of squares of prediction residuals for excluded data. Results of these models for pesticide dissipation in crops were compared with previously used dissipation data in the assessment model framework proposed in Eq. 1 and compared in terms of model input and output uncertainty expressed in terms of GSD^2.

3. Results

Reference half-lives for 333 pesticides with reported temperatures applied at 20°C under field conditions range from 0.2 days for pyrethrins and 0.3 days for chlorsulfuron to 31 days for dalapon with 95% of all half-lives falling in the range between 1 and 18 days. Parameter estimates correct these half-lives for specific crops, temperatures, and study conditions. On average over all substances, temperature imputation only contributes with 5.1% to standard errors of half-lives. Standard errors of substance and plant parameter estimates approximately follow a decrease proportional to the inverse of the square root of the number of data points. With a minimum of 20 data points, we get a standard error of 0.08, implying a 95% confidence interval on geomean half-lives of a factor 1.5. For pesticides without reported data, we developed our final predictive model aims at estimating dissipation half-lives of pesticides as a function of their chemical class and properties. Substance class, plant species, cold storage conditions, temperature, substance molecular weight, octanol/water partition coefficient and saturation vapor pressure were taken into account as final predictor variables. All other tested substance properties (air/water and soil organic carbon/water partition coefficients, half-lives in air and soil) either did not significantly improve model accuracy or showed strong correlation with another variable. This model yields standard errors on the parameter estimates of 0.07 to 0.16 for substance classes and 0.06 to 0.15 for plant species. Implementing the new estimated dissipation half-lives into the model assessment framework proposed in Eq. 1 yields a reduced model input uncertainty in terms of a reduced $GSD^2(hF_{i,c}) = 12$, which corresponds to a reduction in variance of residues in crop harvest across pesticides and crops. Thereby, we reduced overall output-related uncertainty in terms of a reduced $GSD^2(IS_{i,c}) = 278.3$ and the contribution of residues in crop harvest (represented by $hF_{i,c}$) from initially 29.6% to now 19.5% (see Figure 1). With this, we reduced uncertainty in characterizing human exposure to pesticides via food crop consumption.

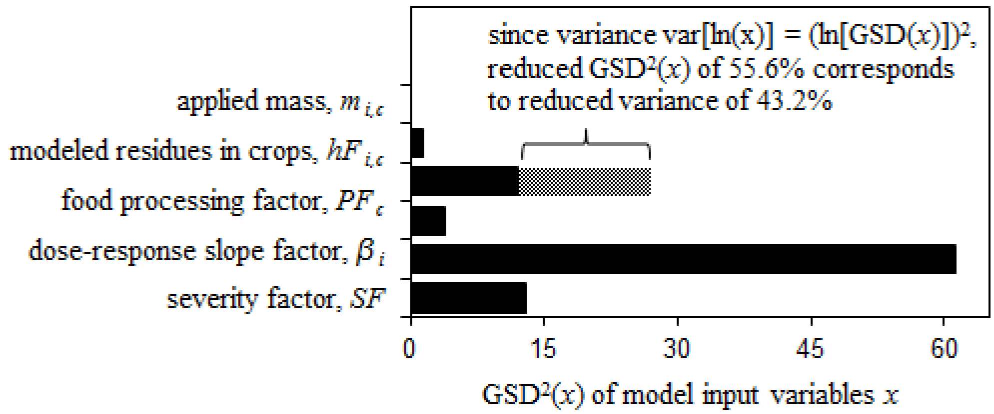

Figure 1. $GSD^2(x)$ of input variables along the pathway from pesticide application to severity of health impacts to obtain impact scores (DALY/year) due to the use of pesticides in EU24 in 2003.

Human health impact scores for use of pesticides in the EU24 countries in 2003 as applied in Fantke et al. (2012a) were re-calculated based on the newly obtained dissipation data. Overall, pesticides contributed annually to 1950 DALY/year in EU24 in 2003, to which only 13 substances applied to grapes/vines, fruit trees, and vegetables accounted for 90% of total annual health impacts. Impact scores thereby range from 0.34 DALY/year for sugar beet and 1.35 DALY/year for cereals to 724 DALY/year for grapes/vines and 1100 DALY/year for vegetables (Figure 2). The total burden per person in hours lost over lifetime was 1.5. Compared with figures for the burden from exposure to fine particulate matter in the air with 195 days or second-hand smoke in the air with 24 days (Hänninen et al. 2014), this figure is relatively low. However, uncertainties in our results – even though reduced for dissipation data in food crops – highlight the figure could be somewhat higher, ranking pesticides with other important environmental stressors in health impact terms when comparing with our upper end 95% confidence interval limit of 17.4 days burden per person over lifetime (see Figure 2).

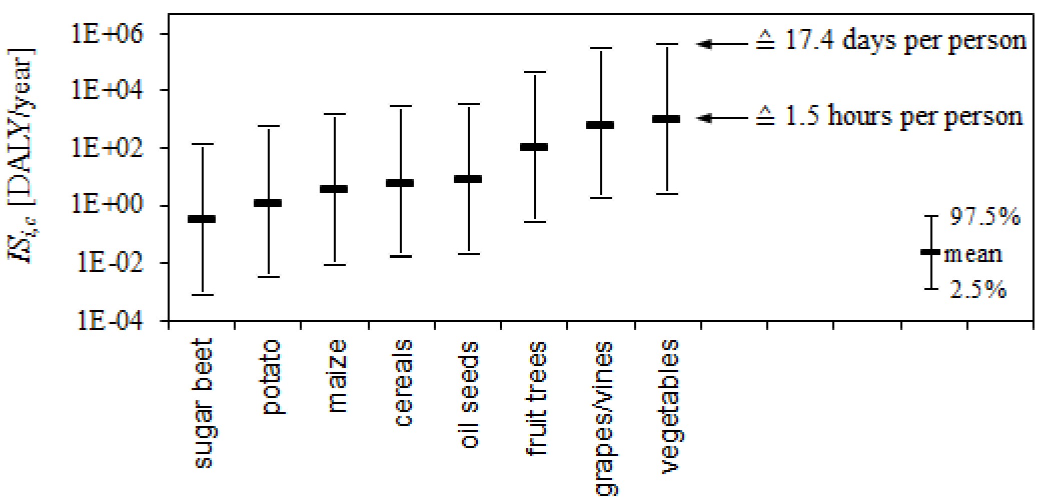

Figure 2. Mean and 95% confidence interval of impact scores (DALY/year) due to the five most extensively used pesticides on each crop class as well as sum over crops across EU24 in 2003.

4. Discussion

We developed a set of 333 comparative pesticide dissipation half-lives in food crops under reference conditions represented by an average plant, field conditions, and a temperature of 20°C (Fantke et al. 2014). These half-lives have been corrected to account for crop type, conditions, and temperature in the underlying experimental studies. This comprehensive approach, where we correct for measured temperature, reported crop species, and reported study conditions for predicting dissipation, and where we impute missing temperatures and finally study the variability of reported half-lives for each pesticide significantly reduces the uncertainty around each estimated dissipation half-life, thereby much better reflecting reference study conditions. Our additional predictive model is designed to estimate dissipation half-lives from properties of individual pesticides belonging to 14 substance classes, for which no half-lives representing reference conditions could be obtained in our study due to missing experimental data.

Considering the high variability between substances and crops, and data availability, our recommended reference half-lives along with the predictive model estimates constitute a first step towards reducing uncertainty in our assessment framework with respect to pesticide degradation in food crops. However, we acknowledge that for LCA studies, the underlying data also need to be available across pesticides and crops. Therefore, additional research is required to systematically assess the relationship between overall pesticide dissipation and degradation in crops, since typically degradation is considered as separate process in LCIA models, to further understand the influence and importance of degradation in crops as one of the main drivers of subsequent human exposure and related health impacts to be included in future LCA studies. Furthermore, reporting guidelines for measuring dissipation from food crops need to be improved with respect to providing sufficient information on environmental study conditions (e.g. reporting temperature, humidity, soil type) and residues in crops (providing enough data points to account for measurement variability and to effectively perform curve fitting for estimating dissipation kinetics). This will further increase LCIA input data quality when using experimentally-derived half-lives.

5. Conclusion

Our pesticide- and crop-specific assessment framework allows for detailed explorations of dietary choices in a LCA context with respect to human health impacts from pesticide exposure via food crop consumption. Furthermore, we reduce uncertainty related to one of the key drivers of pesticide residue dynamics in food crops in support of improving the reliability of impact assessment results. The next step is to include human exposure via crop consumption along with improved pesticide dissipation data into existing LCIA methodologies for consideration in future LCA studies. We further acknowledge that impacts from pesticides in LCAs are often dominated by impacts on biodiversity (Geiger et al. 2010) and on groundwater (Arias-Estévez et al. 2008). Hence, we emphasize the need to include pesticides into food-related LCA studies and recommend covering all relevant impact categories related to pesticides including human toxicity, ecosystem toxicity and groundwater contamination to provide an improved basis for LCA-based decision support.

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Life cycle assessment of Brazilian cashew

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ABSTRACT

Cashew (*Anacardium occidentale*) is a native Brazilian perennial tree that is also present on 30 other tropical countries and produces three worldwide-commercialized food products: cashew nuts, apple, and gum. This study assesses the life cycle environmental impacts of Brazilian dwarf cashew considering the management and crop functions. From the joint analysis of both functional units, we conclude that the best option to improve the environmental performance of Brazilian cashew production is to adopt the reference system with modifications regarding fertilization and land use change. From this case study, we highlight the benefits of considering both functional units and all production stages in the study of perennial crops.

Keywords: cashew nut, cashew apple, cashew gum, dwarf cashew, *Anacardium occidentale*

1. Introduction

Cashew orchards are of great importance for the social economy of many developing regions of the world (Azam-Ali and Judge 2004, Hall et al 2007), since this plant grows in harsh semi-arid climatic and soil conditions and generate income to many small farmers. Cashew producers may benefit from the commercialization of cashew nuts (CN), cashew apple (CA) and gum (CG) from the cashew tree in the international market. The renewal of orchards after 20 years of production also generates wood (CW) that is usually used by potteries as a renewable fuel.

World demand for cashew nuts has increased at a rate of about 4% annually from 2007 to 2011 (FAO 2013, INC 2013). This demand was mostly (79%) attended in 2011 by small farmers located in Vietnam, Nigeria, India, Ivory Coast and Brazil. In 2011, the United States, the United Kingdom, Germany and The Netherlands imported 76% of the cashew nut produced in tropical developing regions.

Cashew agriculture research in Brazil and in other countries has focused in developing high productive cashew clones and production systems. As this goal has been achieved in Brazil, research starts to focus on the evaluation of the environmental sustainability of cashew production systems. The knowledge about the environmental hotspots in cashew production may shed light on the identification of improvement options that will contribute to the environmental sustainability of producing regions. Besides, consumers and companies all over the world utilize cashew nuts in nature or as ingredient in diverse food (e.g. snacks, sauces, protein concentrates), making the cashew nut life cycle assessment of interest for many production chains.

This study assesses the life cycle environmental impacts of Brazilian dwarf cashew (Anacardium *occidentale*). We consider two agriculture functions: land management (agriculture system level) and production (product level). The study considering the land management function evaluates the environmental impacts of two production systems and the study focused on production, the impacts caused by cashew products (CN, CA, CG and CW).

2. Methods

We followed ISO 14040 directives to perform a life cycle assessment (LCA) for quantifying the environmental impacts of Brazilian dwarf cashew systems and products (CN, CA, CG and CW).

2.1. Scope and functional unit

The scope of this study is cradle to cashew farm exit-gate, considering the production of inputs (diesel, fertilizers and pesticides), transport of inputs to the cashew farm, and cultivation of dwarf cashew trees. Two functional units are used: one ha with 208 trees for the study of the cropping system; and one kg for the analysis of cashew products (CN, CA, CG and CW). The inventory data for the study of cropping systems covers the production of dwarf cashew cultivated during 20 years. Cultivation is organized by production stage: nursery, establishment (includes land preparation and planting during the first year), low production (4 years) and full production (years 6 to 20).

2.2. Inventory data

Two production systems for dwarf cashew are considered: the reference system (REF-farm) developed along 20 years of research, and the low input system (LI-farm) as practiced by a sample of farms.

Input data for the reference system come from reports of experiments and interviews with researchers on the subject of dwarf cashew production systems. Experiments started in 1980 at the Embrapa Tropical Agroindustry Experimental farm, located at Pacajus, Ceará State, Brazil. Their results are summarized by Crisóstomo et al (2007), Cardoso et al (2013), Crisóstomo (2013), Cavalcanti Junior (2013), Mesquita and Sobrinho (2013), Miranda (2013), Serrano and Oliveira (2013), and Vidal Neto et al (2013).

Input data for the low input system is derived from interviews applied to a sample of ten cashew farmers and two rural extension managers in 2013, located in six municipalities of the Ceará State, Brazil. The Ceará State holds 57% of the cashew production area (IBGE, 2013). The interviewed farmers and extensionists provided the average amount of input and production obtained for CN and CA during production stages, considering the last three years.

Inventory data for the cradle to gate production and transport of inputs used on cashew orchards (manure, mineral fertilizers, diesel, pesticides and plastics) were obtained from the *ecoinvent* 3.0 database (Weidema et al 2013).

2.3. Description of dwarf cashew production

Dwarf cashew production in the REF and LI farms encompass the following stages:

Nursery: Cashew seedlings are produced by grafting in nursery farms. The nursery involves three steps that take 120 to 150 days. First, dwarf seeds are sown on compost substrates that are deposited in small polypropylene tubes (diameter of 6 cm and height of 20). After 60 days of sowing, the germinated plants are grafted on rootstocks of the scion obtained from dwarf clone gardens. The grafted plants stay during 30 to 40 days in a greenhouse. After this time, seedlings are acclimatized in open fields for 30 to 50 days when they are apt to be planted in cashew orchards. Irrigation of the seedlings occurs every day of the production cycle and foliar fertilization, four times per cycle. We considered that 60,000 seedlings of grafted cashew are produced per year in 5.6 ha, with 60% of seeds producing viable seedlings; 20x6x4m grille of high density polyethylene are used to cover the shaded greenhouse (50%), lasting for 5 years; and that the polypropylene tubes last for 10 years.

Establishment (year 1): Land conversion, plowing and harrowing are performed before seedlings transplantation to orchards. The transplantation occurs in the raining season (January to April). Grafted cashew seedlings are planted into pits of 40x40x40cm on sandy soils with the pits spaced 8x6m (208 plants.ha^{-1}). Calcareous dolomite is applied at the bottom of the pits that are then filled with a mixture of surface soil, micronutrients, bovine manure and simple superphosphate at the REF-farm or only with soil poultry or bovine manure at LI-farms. Post-planting fertilization at REF-farm is based on nitrogen and potassium fertilization and at LI-farm on nitrogen from manure. Manual pruning is practiced in order to take out stems, which emerge from the rootstock, and flowers, in order to foster plant growth. In this first year, at the REF-farm, localized irrigation at the pits occurs four times per month during the dry season (from May to December).

Low production (year 2 to 6): Dwarf cashew plants grow until the sixth year after planting when they achieve growth stability. During each production year, the tree has a low and an intense vegetative growth: low in the raining season (January to April) and an intense growth in the dry season (May to December). During the intense growth period, the leaves fall, and flowering and fructification occur. Cashew orchards are rain fed from the second year onwards. Fertilization aiming plant growth occurs along the raining season (January to April). REF-farm uses manure and mineral (NPK) fertilizers and LI-farm, only manure. Manual pruning is practiced after harvest (December to January) and aims to reduce branches interlacing. This increases illumination in the canopy, and prevents pest and disease dissemination among branches. The REF-farm applies herbicides or mechanical weeding below the tree canopy to prevent competition among cashew and other plants (weeds). The LI-farms use manual weeding only before harvest. Both the REF-farm and LI-farms mow between lines of cashew trees to facilitate harvesting and promote soil covering with green manure (mulch). Insecticides, fungicides and herbicides are used only at REF-farm and follow the integrated pest and disease management plan. As many as 97 species of insects and 10 types of fungus may attack cashew orchards in Brazil. The calculus of the amount of pesticide used, however, consider only those pests and diseases that have caused most damage to orchards and that have registered pesticides for this culture. The production of CN (fruit) and CA (pseudo fruit) starts on the second year, and the CG, on the sixth year. CG is produced with the monthly application of growth hormone on lateral cuts made at the tree trunk.

Full production (year 7 to 20): The adult dwarf cashew tree may be 4 m high with a canopy diameter of 6 to 8 m and roots that may achieve 12 m (Crisóstimo et al 2007). CN (kernel and shell) average weigh ranges from 7 to 12 g, and CA, from 80 to 160g. During the full production period, fertilization aiming the CN production of 1,200 kg.ha^{-1}.year^{-1}, CA of 2,160 kg.ha^{-1}.year^{-1} and CG of 90 kg.ha^{-1}.year^{-1} is applied at the REF-farm. LI-farm relies only on composted manure and may produce 600 kg.ha^{-1}.year^{-1} of CN and 1,080 kg.ha^{-1}.year^{-1} of CA. Annual mowing, weeding and pruning are performed as in the low production years. At the end of the 20th year, new cashew trees replace old ones and the CW is commercialized at the local market.

2.4. Calculation of emissions

The calculus of greenhouse gas emissions (CO_2, CH_4 and N_2O) from agriculture fields for both production systems follows IPCC (2006). The following airborne emissions are estimated according to the methods described by Nemecek and Schnetzer (2012): ammonia (NH_3) and nitrogen oxides (NO_x) to air; nitrate (NO_3) and phosphorous compounds as phosphate (PO_4) and phosphorous (P) to water; pesticides and heavy metals (Cadmium (Cd), copper (Cu), zinc (Zn), lead (Pb), nickel (Ni) and chrome (Cr)) to soil. All pesticides applied for crop production were assumed to end up as emissions to, and remain as pollutants in the soil. The further fate of the substances is taken into account in the impact methods for human toxicity. We assumed that no leaching of soluble phosphate (PO_4) to ground water occurred because of the high aluminum and iron oxides content usually found in the type of soils where cashew orchards are located (Crisóstomo et al 2007) in Brazil. These oxides adsorb phosphate molecules in soil particles impeding leachate.

The humidity, carbon and nitrogen content of dwarf cashew tree are measured according to Silva et al (2009) from samples of roots, trunk, leaves and steams of a ten years old Clone BRS 226 tree. Samples were collected at Embrapa Pacajus Experimental Station, Ceará, Brazil.

2.5. Allocation procedure

No allocation is needed for the functional unit of one ha with 208 trees. Allocation is necessary, however, for the functional unit of one kg cashew nut products due to the multiple cashew products harvested from a dwarf cashew tree (CN, CA, CG and CW). We applied allocation according to mass and economic criteria to illustrate sensitivity of results for the allocation principle used. Allocation considers the production weight of cashew products over 20 years. The replacement of the cashew orchard allows the commercialization of CW at the end of the 20th year. CG may start to be extracted from dwarf cashew by commercial farms in the sixth year of plant growth. CG market value is assumed the same as the acacia gum that is a similar product and has long been commercialized in the world market. The allocation factors for each farm system and product is presented in Table 1.

Table 1. Allocation factors

		REF-farm			LI-farm		
Product	**Price (US$/kg)**	**Production in 20 years (kg/ha)**	**Economic allocation**	**Mass allocation**	**Production in 20 years (kg/ha)**	**Economic allocation**	**Mass allocation**
CN	0.8	17,330.00	44%	15%	6,432.50	51%	8%
CA	0.4	31,194.00	39%	27%	11,578.50	46%	14%
CG	4.0	1,260.00	16%	1%			
CW	0.004	67,392.00	1%	58%	67,392.00	3%	79%
Total		117,176.00	100%	100%	85,403.00	100%	100%

2.6. Impact assessment

We applied the Recipe model (Goedkoop et al 2009) focusing on the midpoint environmental impact categories natural land transformation, climate change, acidification, eutrophication and human toxicity. The reasoning behind this selection is that these categories are important to the study of fruit products, according to Michalopoulos and Christodoulopoulou 2012). Furthermore, the emissions calculated for cashew production in this study may directly cause these impacts.

The reference situation for land transformation is the change from Caatinga (Brazilian savannah) forest to dwarf cashew orchards.

The method Monte Carlo for uncertainty analysis is applied to the study results. We assume log normal distributions for probability functions and run the model for 1,000 cycles. The Pedigree matrix in SimaPro is used to determine the deviations of each variable (Goedkoop et al., 2008).

3. Results

3.1. Inventory of inputs from primary data per ha

The full and low production stages require most of the inputs per hectare of cashew cultivated in both production systems, i.e. REF-farm and LI-farms (Table 2). In these stages, macronutrients and pesticides are more intensively applied than in the earlier stages. Nursery requires in general the smallest amounts of inputs, although the establishment year requires most of the water and all the micronutrients (boron, copper, manganese, molybdenum, zinc and iron).

Table 2. Primary data of dwarf cashew REF and LI farms, per ha, in each production stage

			REF-farm				LI-farm			
Inventory	**Unit**	**Nursery**	**Establish-ment (year 1)**	**Low production (years 2 to 6)**	**Full produc-tion (years 7 to 20)**	**Total value for 20 years**	**Establishment (year 1)**	**Low production (years 2 to 6)**	**Full production (years 7 to 20)**	**Total value for 20 years**
Seedlings	number	208	208			208	208			208
Cashew nut	t			2.93	14.40	17.33		0.73	5.70	6,43
Cashew ap-ple	t			5.27	25.92	31.19		1.32	10.26	11,58
Cashew gum	t			0.09	1.17	1.26				-
Cashew wood	t				67.39	67.39			67.39	9,36
Inputs										
Area transf.	ha	0.03				1.00				1,00
Lime	t		2.01	2.50	6.49	11.00	0.04	0.21	0.05	0.30
Gypsum	kg		500.00			500.00				
Seeds	kg	1.80								
Boron	g		374.40			374.40				
Copper	g		166.40			166.40				
Manganese	g		416.00			416.00				
Molybdenum	g		20.80			20.80				
Zinc	g		1,872.00			1,872.00				
Iron	g		624.00			624.00				
Urea	kg	0.003	21.27	406.55	3,091.64	3,519.45				
potassium chloride	kg	0.003	10.76	208.00	717.24	936.00				
Single super-phosphate	kg	0.003	161.78	1,109.33	2,311.11	3,582.22				
Organic compost	kg		2,080.00	14,560.00	54,080.00	70,720.00	2,496.00	12,480.00	32,448.00	47,424.00
Deltamethrin	kg	0.003	0.01	1.30	6.49	7.80				
Copper oxyc.	kg	0.02	0.40	34.28	171.43	206.11				
Glyphosate	kg			3.68	21.23	24.90				
Ethephon	kg			3.59	46.73	50.32				
Diesel	l		345.25	461.25	1,199.25	2,005.75	80.00	400.00	1,040.00	1,520.00
Water	l	4,659.20	94,900.00			94,900.00	94,900.00			94,900.00
Plastic	g	213.40								

The majority of dwarf cashew orchards are located on deep, well-drained, sandy loam soil of low natural fertility, usually Quartzarenic Neosols (Quartz Sands), Latosols and Argisols (Podzolics) (Crisóstomo et al 2007). Fertilization is required in these soils to promote plant growth and improve yield. Potassium and phosphorous are essential during growth (low production) stage for root development. Nitrogen increases the flowering period and the yields of CN, CA and CG (Crisóstomo 2013, Lima 2014). The LI-farms apply per ha less fertilizer than the REF-farm in all production stages.

Pesticides are required to control the population of pests and diseases when infestation levels are above the rates determined by the integrated management plan (Cardoso et al 2013, Mesquista et al 2013). These pests and diseases attack seedlings and cashew trees during inflorescence, reducing nut and apple yields. Only the REF-farm controls infestations of pests and diseases. In this farm, deltamethrin (insecticide) is used to control infestations of drills (*Anthistarcha binocularis Meyrick*) and thrips (*Selenothrips rubrocinctus Giard*), and copper oxychlorate (fungicide), to combat Anthracnose (*Colletotrichum gloeosporioides (Penz) Pez. & Sacc)*. Glyphosate (herbicide) is applied to control weeds, especially *Senna obtusifolia, Alternanthera sp., Panicum maximum, Cenchrus echinatus*, *Eleusine indica, Setaria geniculate, Ipomoea sp., Sida cordifolia, and Solanum sp.*

The low fertilization and no control of pests and diseases in LI-farms result in CN and CA yields that per ha are 63% lower than in REF-farm. CG is a new product from cashew orchards and is not yet produced by most of the commercial farms in Brazil. Nonetheless, Lima (2014) reports that cashew orchards with reduced fertilization rates (150 g.plant^{-1} of N, 300 g.plant^{-1} of P_2O_5 and 90 g.plant^{-1} of K_2O) produce 7% less CG yield per ha than orchards subjected to fertilization as adopted in the REF-farm.

Lime and gypsum are applied to cashew orchard soils, both in the REF and LI-system, because of their high acidity and aluminum saturation percentage (above 40%) (Crisóstomo 2013). Gypsum is incorporated in deep soil layers to reduce aluminum saturation before seedling transplantation at the REF-farm, while lime is applied on all production stages to correct soil pH in both farm systems.

The best climate conditions for dwarf cashew production occur when temperature ranges from 22 to 32^{o}C along the year, relative humidity, from 70% to 85%, and annual precipitation, from 800 to 1,500 mm, distributed over 5 to 7 months per year (Serrano and Oliveira, 2013). The presence of this climate condition allows cashew cultivation in semi-arid regions without irrigation, such as the Brazilian Northeast. Nonetheless, during nursery and the first establishment year, continuous water supply through irrigation is necessary and used by REF and LI-farms. Irrigation is necessary because of the small size of the rooting system that is not able yet to extract water from deep soil layers.

3.2. Impacts per ha of dwarf cashew considering the land management function

The production system adopted by LI-farms cause significant less environmental impacts per ha than the system used by the REF-farm, for all but the land use impact category (Table 3). This result is expected since the LI-farms use less input and generate less emission than the REF-farm. However, both systems may require land transformation for the same area.

The main sources of environmental impacts at the REF-farm and LI-farm are:

Land use and climate change: Carbon stock losses during land transformation and nitrous oxide emissions from nitrogen fertilizers used in cashew orchards emit most of the greenhouse gases causing climate change.

Human toxicity: Fertilizers are contaminated by heavy metals and these are released to soil (e.g. Cd when single superphosphate is applied to agriculture soil). Pesticides applied in cashew orchards are also released to soil. These emissions may degrade human health. This occurs in the REF-farm that uses mineral fertilizers and pesticides. In LI-farms, upstream processes related to the poultry manure production chain emit most of toxic substances.

Terrestrial acidification: The use of mineral nitrogen fertilizers at the REF-farm emits most of the NH_3 and NO_x that cause terrestrial acidification. For the LI-farms, NH_3 emitted in the production of composted poultry manure is the main cause of acidification.

Marine and freshwater eutrophication: The production and use of mineral and organic fertilizers on both REF and LI farms emit most of the nitrogen and phosphorous compounds with potential to cause marine and freshwater eutrophication.

Table 3. Environmental impacts per ha in REF and LI farms, for 20 years

Impact category	REF-farm	Li-farm	Uncertainty (LI-farm >= REF-farm), Confidence interval of 95%
Climate change (kg CO_2 eq.ha^{-1})	176,556.95	117,300.62	1%
Human toxicity (kg 1,4-DB eq.ha^{-1})	36,890.62	768.44	0%
Terrestrial acidification (kg SO_2 eq.ha^{-1})	1,426.53	43.40	0%
Freshwater eutrophication (kg P eq.ha^{-1})	20.22	2.72	0%
Marine eutrophication (kg N eq.ha^{-1})	258.68	142.15	0%
Natural land transformation (m^2.ha^{-1})	19,984.02	19,972.88	51%

3.3. Impacts per kg of cashew products considering the crop production function

Mass as well as economic allocation is used to calculate the environmental impact per kg cashew product (see section 2.5 and Table 1 for mass and economic values). The type of allocation criteria affects the magnitude of cashew products impacts (Table 3). CG is the most sensible product to the choice of allocation criteria due to its relative high economic value. The impact values for CG may increase at least fifteen times when moving from mass to economic allocation.

Table 4. Impacts per kg of CN, CA, CG and CW, considering economic and mass allocation

Cashew Product	System	Allocation procedure	Climate change (kg CO2 eq.kg^{-1})	Terrestrial acidification (kg SO2 eq.kg^{-1})	Freshwater eutrophication (kg P eq.kg^{-1})	Marine eutrophication (kg N eq.kg^{-1})	Human toxicity (kg 1,4-DB eq.kg^{-1})	Land transf. (m2.kg^{-1})
CN	REF-farm	Economic	4.46	0.04	0.0005	0.01	0.93	0.51
		Mass	1.51	0.01	0.0002	0.002	0.315	0.17
	LI-farm	Economic	9.34	0.003	0.0002	0.01	0.06	1.59
		Mass	1.37	0.001	0.00003	0.002	0.009	0.23
CA	REF-farm	Economic	2.23	0.02	0.0003	0.003	0.47	0.25
		Mass	1.51	0.01	0.0002	0.002	0.31	0.17
	LI-farm	Economic	4.67	0.002	0.0001	0.006	0.03	0.80
		Mass	1.37	0.001	0.00003	0.002	0.009	0.23
CG	REF-farm	Economic	22.31	0.18	0.0026	0.033	4.66	2.53
		Mass	1.51	0.01	0.0002	0.002	0.31	0.17
	LI-farm	Economic						
		Mass						
CW	REF-farm	Economic	0.02	0.0002	0.000003	0.00003	0.005	0.003
		Mass	1.51	0.01	0.0002	0.002	0.31	0.17
	LI-farm	Economic	0.05	0.00002	0.000001	0.0001	0.0003	0.01
		Mass	1.37	0.0005	0.00003	0.002	0.009	0.23

Economic allocation assigns higher impacts for CN, CA and CG in both farm systems since these products have higher market values. Since CG is not yet widely commercialized by LI-farms, no impacts are assigned to this product for this system.

Moreover, mass allocation highlights the participation of CW in impact values, in both farm systems. The lower CN and CA yields obtained in LI-farms explain the higher allocation value assigned for CW.

The combined analysis of allocation criteria and production systems shows that the environmental performance of products cultivated in different production systems change according to the allocation criteria adopted.

If the choice is for economic allocation, CN, CA and CW cause less impact on land use, marine eutrophication and climate change when produced in the REF-farm. If mass allocation is used, these products cause similar impacts for these impact categories. The high yields obtained in the REF-farm are not enough to reduce the impacts of cashew products on the other impact categories (freshwater eutrophication, human toxicity and acidification) to values lower to the ones obtained in the LI-farm.

4. Discussion

In this study, cashew production was evaluated at the per ha (land management) and per kg product (production) levels. The land management study compared two production systems: one resulting from consistent field research (REF-farm system) and the other based on common practice at small and medium sized farms (LI-farm). The comparison of these systems showed that LI-farms perform better in terms of environmental impacts per ha.

However, these farms maintained the same low fertilization rates in all production stages, and hardly controlled pests and diseases, leading to low emissions and impacts but also low yields. These low yields of CA and CN and no production of CG may result in higher impact per kg of product for LI-farms than for ref-farms, according to the allocation procedure chosen. The low yields from the LI-farms also mean low incomes for the small farmers, reducing their capacity to invest in better management practices. The interviews with the LI-farmers also highlighted the reduced access to technical assistance and little knowledge of updated cashew production systems. Considering these aspects, the argument for refraining from the LI-farm system becomes stronger. A better option may be to keep the high yields of the REF-farm system and try to reduce its environmental impacts per ha.

Three main activities considerably affect the environmental performance of the REF-farm system: land use transformation, mineral fertilization and pesticide applications. The best choice for reducing impacts on land use, and as a consequence on climate change and biodiversity, is to establish dwarf cashew orchards in already deforested areas currently set aside or used to produce annual crops. Conversion of Caatinga (Savana) vegetation to dwarf cashew orchards results in losses of carbon stocked on biomass and soil and also cause biodiversity loss in the already much disturbed Caatinga biome. Regarding fertilization options, organic instead of mineral fertilization is preferred from the perspective of impacts per ha. The use of mineral fertilizers in the REF-farm is responsible for the major impacts on human toxicity, acidification and eutrophication. The amount of pesticides can be decreased with the use of intercrop systems that increase biodiversity in cashew orchards (Xavier et al 2013).

The intercrop of cashew trees with leguminous and grass species may decrease the need for mineral fertilization and reduce the level of pest and diseases in orchards. Some small cashew farmers in the Brazilian Northeast already adopt this practice to increase their sources of income, cultivating cashew in association with beans and corn during the raining season, but medium and large farms hardly adopt this practice. Previous research showed that the cultivation of *Canavalia ensiformes* (type of bean) between lines of dwarf cashew trees during the raining season leveled CN yield to 1,179 $kg.ha^{-1}$ already in the fourth low production year (Oliveira et al 2000). This yield is 108.7% higher than the one obtained in the experimental parcel with no consociation. The association of legumes with cashew also reduces weed development and the need for the use of herbicides, besides increasing soil carbon stock, organic matter and nutrient content (Xavier et al 2013). However, further research is still necessary for determining the amount of mineral fertilizers that can be substituted by green fertilization with leguminous species adapted for the semi-arid environmental conditions.

The impact per kg product level was evaluated by means of a sensitivity analysis with both economic and mass criteria to allocate impacts over the multiple product outputs of the cashew orchards. Although both allocation procedures can be adopted to calculate the impacts of cashew products, results considerably change with the choice made. Economic allocation increases the impacts of CN, CA and CG and decreases the impact of CW in all categories. It also leads to the conclusion that products from the REF-farm may cause lower impacts than when cultivated in LI-farms, according to the analyzed impact category. On the other hand, mass allocation does not differentiate products from both production systems, for half of the analyzed impacts categories.

CG is presently starting to be extracted from dwarf cashew by commercial farms and the assumptions made in this study for the evaluation as CG using mass allocation represents a near future situation for Brazilian cashew farms. Nonetheless, CG real price is not settled yet in the local or international markets. The value used for CG was assumed the same as the acacia gum, although this situation is not yet at stake now since CG is starting to be commercialized. The acceptance of this new product by the food industry is not yet clear and it may take years for CG to become a real substitute product for the acacia gum. Nevertheless, CG is expected to be commercialized in few years, and the study results are therewith representing a future situation.

The prices for CN and CA are volatile ranging from $ 0.4 to 0.8 per kg of CN and from $ 0.2 to 0.4 per kg of CA, respectively, in the last five years. We therefore recommend the use of mass allocation for the evaluation of cashew products. This is also in accordance with ISO 14040 that recommends allocation on physical relationships over allocation based on other relationships.

5. Conclusion

Cashew production in small and medium farms in Brazil use a low input system with reduced environmental impacts per ha. These farms, however, also have low yields and profits. Many years of experimental research developed a reference production system with increased yields of nuts and apples as well as gums extracted from the trees trunk. This reference system relies on fertilization with specific rates for each production stage, and on the integrated management of pests and diseases. The higher yields per ha result in the reference system having similar or lower impacts per kg of product than the low input systems, for half of the impact categories.

The impacts caused by Brazilian cashew orchards and products change according to the choice of functional unit (i.e. per ha or per kg product) and allocation procedure adopted (i.e. mass or economic allocation). When the land management function is applied (one ha of cultivated orchard), the low input production system is preferable, but if the production function is chosen the results per kg of product produced in each system are not conclusive.

From the joint analysis of both functional units, we conclude that the best option to improve the environmental performance of Brazilian cashew production is to adopt the REF-farm system with modifications. That is, this system provides higher yields and income for the cashew farmers and may cause less impact if orchards are installed in agricultural or degraded areas and intercrop practices are adopted.

6. Acknowledgement

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Contemporary comparative LCA of commercial farming and urban agriculture for selected fresh vegetables consumed in Denver, Colorado

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Keywords: comparative LCA, emissions reporting, urban agriculture, land use change, fresh vegetable production

ABSTRACT

Local policy makers typically do not have useful, quantitative metrics to compare environmental costs and benefits of urban vegetable production versus the large-scale commercial production in the typical grocery store supply chain. While urban agriculture has been championed as a way to address social issues such as food access and nutrition, we know relatively little about net environmental benefits, if any. The study combines a comparative life cycle assessment of vegetables with effects of direct and indirect land use change resulting from the urban vegetable production. This paper presents a methodology and selected results of scenarios of land use change due to urban vegetable production address resource use, greenhouse gas emissions, employment, and soil organic carbon. Surprisingly, urban vegetable production is not categorically favorable for each metric; several key parameters can shift the balance in favor or out of favor for either growing format, and these parameters are distinctly bottom-up.

1. Introduction

The popularity of urban vegetable production on a small-scale is, in part, explained by the way people feel about the food system, and is a response of sorts – a response to the existing dominance of large-scale production formats. Much of what we know about the supposed benefits of small-scale production, then, is relative to large-scale production. Typically, the belief is that small-scale is better than large-scale. However, a quantitative comparison of these two formats using a life cycle approach that could corroborate these beliefs has not been done on a scale that is relevant. Further, to have relevance with a number of municipal climate and sustainability action plans, a life cycle assessment would have to be fairly specific to that locale.

Food life cycle assessment (LCA) studies commonly draw from any one or more of the intermediate sources, from raw data to proprietary software tools, as depicted in Figure 1. Pathways of agricultural production emissions reporting can include measured, estimated, and modeled data and each use can support far-reaching and impactful policy decisions on the part of government and industry (Grubb 1995; Lynch et al. 2011; Smith et al. 2008; Zborel et al. 2012). This study carefully chooses appropriate data sources that match the level of variability and scale of the components of the vegetable production system where national or regional data may be used otherwise. In doing so, methodological accuracy is increased and uncertainty is reduced, although a component of this study relies on data collected in a case study whose sample size is small. This paper presents selected data, methodology, and findings from a more comprehensive study (Fisher 2014).

While some LCAs are conducted with interest in greenhouse gas accounting, this study contributes additional valuable environmental metrics such as resource use (for water, energy, fuel), ecotoxicity, and human health impacts. Additional impact metrics such as soil organic carbon (SOC) levels, indicating soil health and supporting ecosystem services, land use change, and employment provide a larger picture of sustainability. SOC is interesting for carbon sequestration and indicates if soil is being improved or depleted. Second, the labor indicator is used because there is a stark and measurable difference between the labor inputs of large-scale and small-scale growing. This can be seen as an opportunity for local employment, but a risk for corporate efficiency.

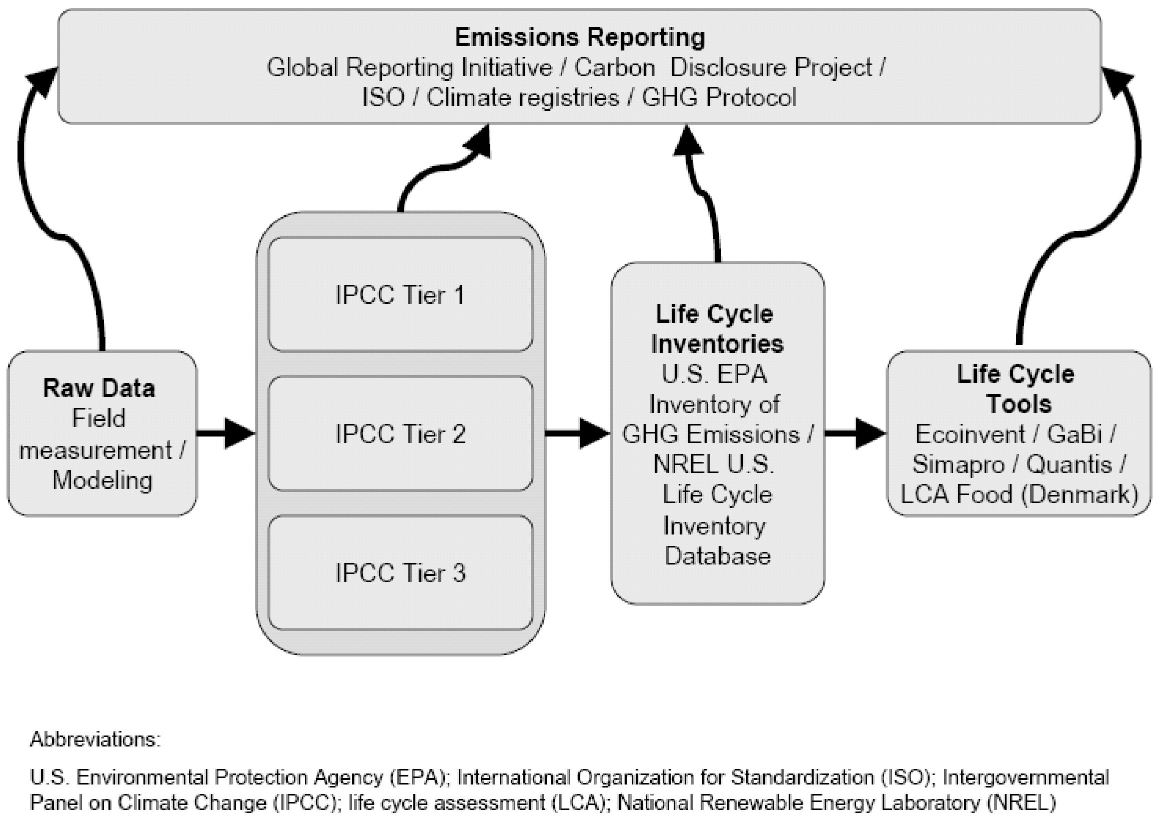

Figure 1. Data Sources and Emissions Reporting.

2. Methods

Formally speaking, this study is an attributional LCA with a component of consequential LCA using direct and indirect land use scenarios (decisions). Although mostly a process-based LCA, a few flows have been characterized by economic input-output and databases that represent regions and nations. For this component, the LCA study could be characterized as hybrid. The LCA follows the well-recognized International Standards Organisation (ISO) Series 14000 standards.

2.1. Scope and System Components

A group of four vegetables was chosen based on their contribution, by calorie, of the average American diet of fresh vegetables. These are potato 53 percent; onion 12 percent; carrot 7 percent; and tomato 6 percent. These comprise approximately 78 percent of the fresh vegetable consumption of the average American (USDA 2009). These are found readily in any supermarket, and are all grown in the Denver metropolitan area by operators of neighborhood supported agriculture (NSA), community supported agriculture (CSA), and individuals. The LCA was conducted on these four vegetables, from field to consumer, in each of nine locations where they are grown. The small-scale urban production occurred entirely in Denver, Colorado. The large-scale commercial production occurred in eight different locations (Comazzi 2012), as shown on Figure 2. The functional unit was set to 1 pound of fresh vegetable (potato, onion, carrot, tomato) to the consumer.

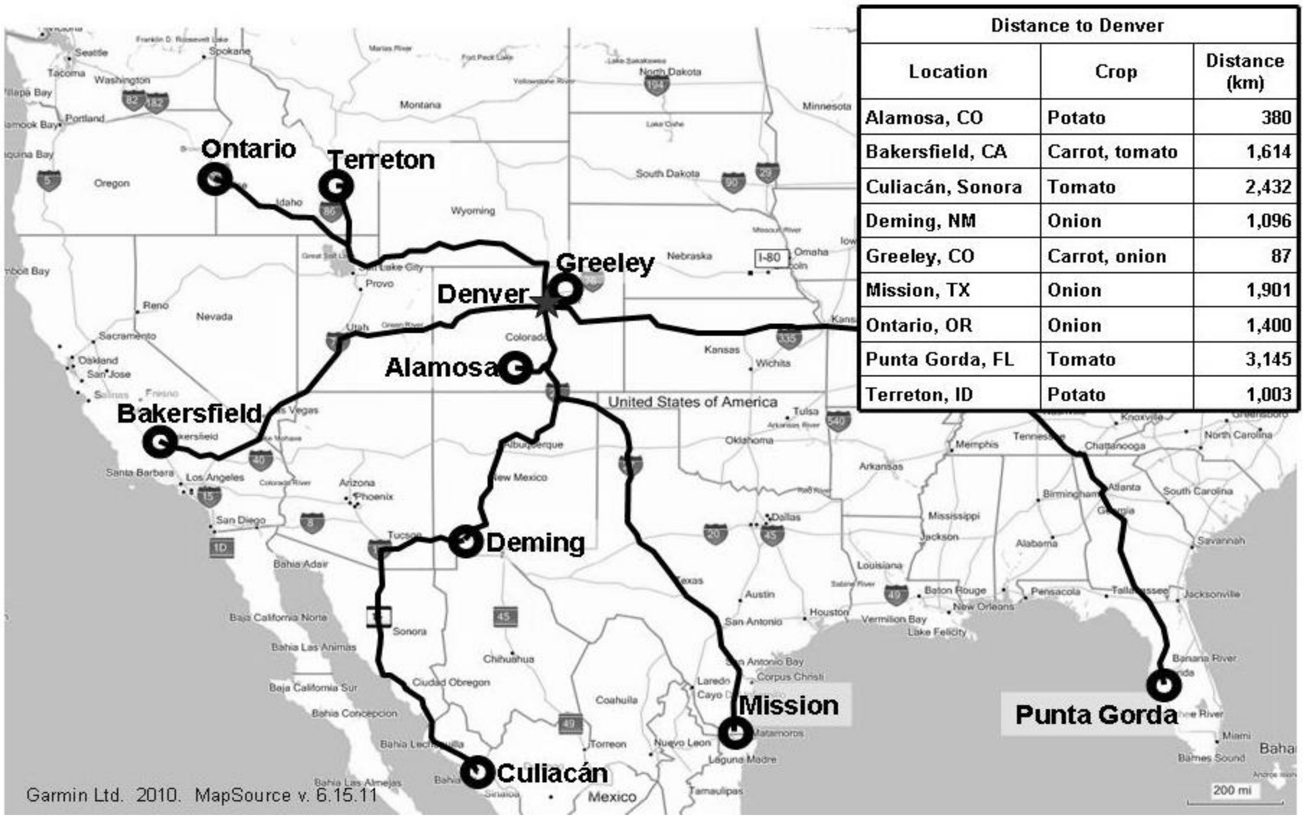

Figure 2. Regional sources of supermarket produce for Denver, Colorado.

2.2. Inventory

Life cycle inventory materials and flows were derived from published sources, such as agricultural extension services farm enterprise budgets, phone interviews with extension agents, and direct measurement (in the case of four growing operations in Denver, Colorado). Most of the inventory involved materials that were assumed to have properties that do not vary with location (e.g., fuel, plastics, fertilizer, etc.). However, a key contribution of this study is recognizing that the plant/soil ecosystem is influenced greatly by site-specific factors. For this reason, the inventory, components supporting ecosystem services (e.g., SOC, N-fixation), and impacts for soil and agricultural production were characterized using the Denitrification-Decomposition (DNDC) model (Li 2000). The inventory list and data sources are presented in Table 1 and Figure 3, respectively.

Table 1. Life cycle inventory materials.

Category	Specific Material	Region	Unit
Fossil Fuel	Diesel, gasoline (well to pump – WTP)	U.S.	L
Electricity	Electricity (Scope 1,2)	eGrid region	kWh
Soil	Bagged potting soil – organic	U.S.	kg
Fertilizer	N, P, K, Zn, Mn, Mg, Cu, gypsum, sulfur, lime	U.S.	kg
Plastics (virgin resins)	HDPE, LDPE, PP, PS, PET	U.S.	kg
Paper	Cardboard	U.S.	kg
Chemicals	Herbicide, pesticide, fungicide	U.S.	kg
Water	Raw irrigation water	Site-specific	L
Water	Potable irrigation water	Denver, CO	L
Transport	Refrigerated tractor / 17-ton trailer	Varies	km
Transport	Light pickup truck	Denver, CO	km
Web Hosting	20 Mb site, 1 yr.	U.S.	year

A key parameter directly tied to the functional unit is yield. Typical yields for the four vegetables under both growing formats, used as the basis of the functional unit, are presented in Table 2. Because the sample size was low for the urban case studies, it was recognized that yield values from direct measurements during only one growing season should be modified based on expert interviews and a sense of what is typical.

Table 2. Estimated yields

Crop	Estimated Typical Yield – Commercial	Estimated Typical Yield – Urban
Potato	1.15 (lbs/ft^2)	0.88 (lbs/ft^2)
Carrot	2.02 (lbs/ft^2)	0.94 (lbs/ft^2)
Onion	0.60 (lbs/ft^2)	0.74 (lbs/ft^2)
Tomato	0.60 (lbs/ft^2)	0.77 (lbs/ft^2)
Notes: Commercial yields adapted and summarized from enterprise budgets published by land grant university cooperative extension services in respective crop regions. Urban yields estimated from expert interview and direct measurement from the case studies.		

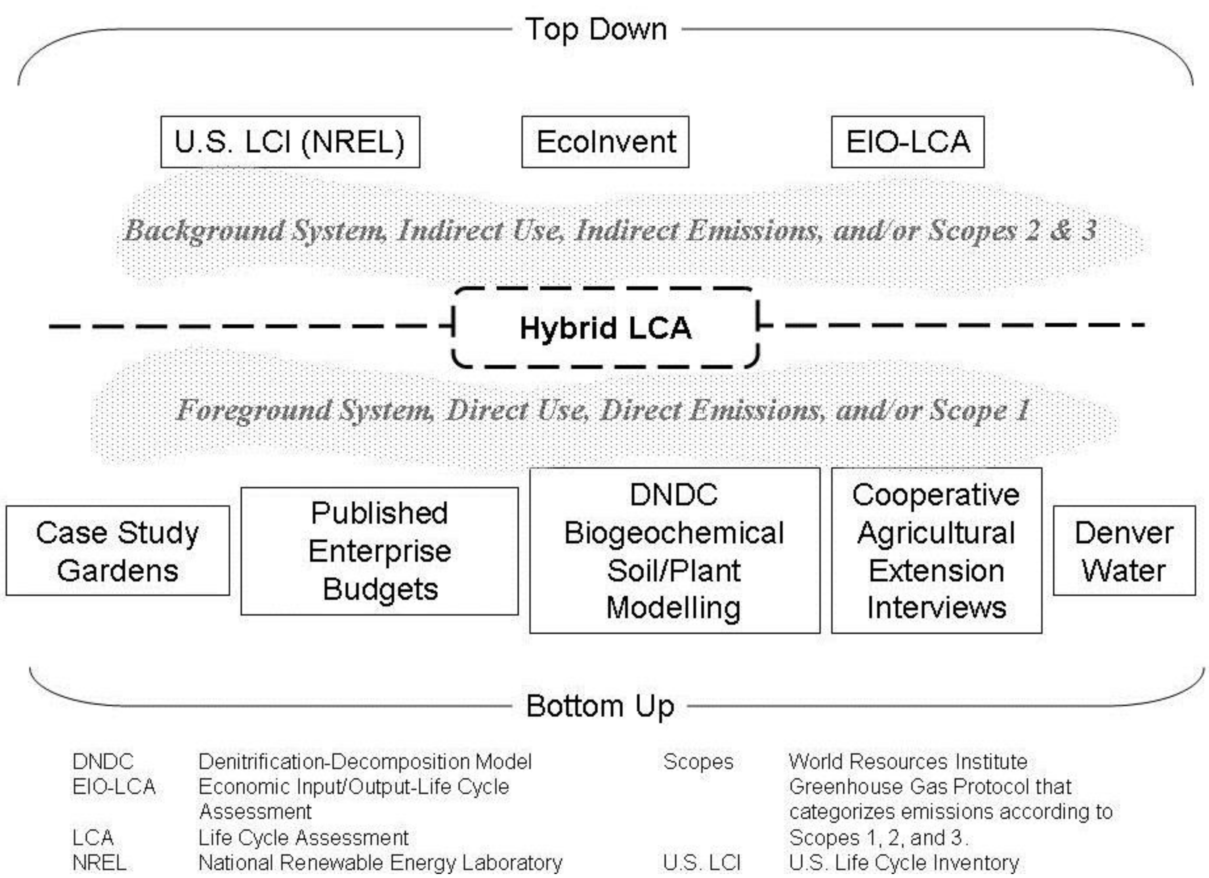

Figure 3. LCA Data Sources.

2.3. Land Use Change

It is asserted that land area used for vegetable production in one location is put into service or out of service based on the incidence of vegetable production in another location, normalized by respective yields. In this way, the product (vegetable) retains an embedded transboundary supply chain accounting element, even though vegetables are grown or consumed in only one location. A number of researchers have characterized SOC and agricultural emissions for various long-term land uses and as it relates to land use change (Kim 2009; Pouyat et al. 2007). This study expands the inventory and life cycle impact categories more comprehensively and sets urban gardening as a type of land use that displaces existing land uses. Conversion to urban garden involves displacement from one of three land uses, namely:

1. Large-scale commercial farmland – each new instance of urban gardening is assumed to displace an equal amount of commercial farmland, adjusted for differences in land productivity related to the functional unit.
2. Neglected and degraded urban areas – in the existing urban setting, a new urban garden can purpose these unused areas that typically have the poorest soil health and lowest SOC levels.
3. Residential turf grass – in the existing urban setting, the demand for space for an urban garden may force conversion from turf grass.

These land uses and scenarios are shown on Figure 4. Although common land uses exist in the urban setting, they are assumed to be similar in the degree of dense plant matter, cultivation, and richly-maintained soil that urban vegetable gardening has.

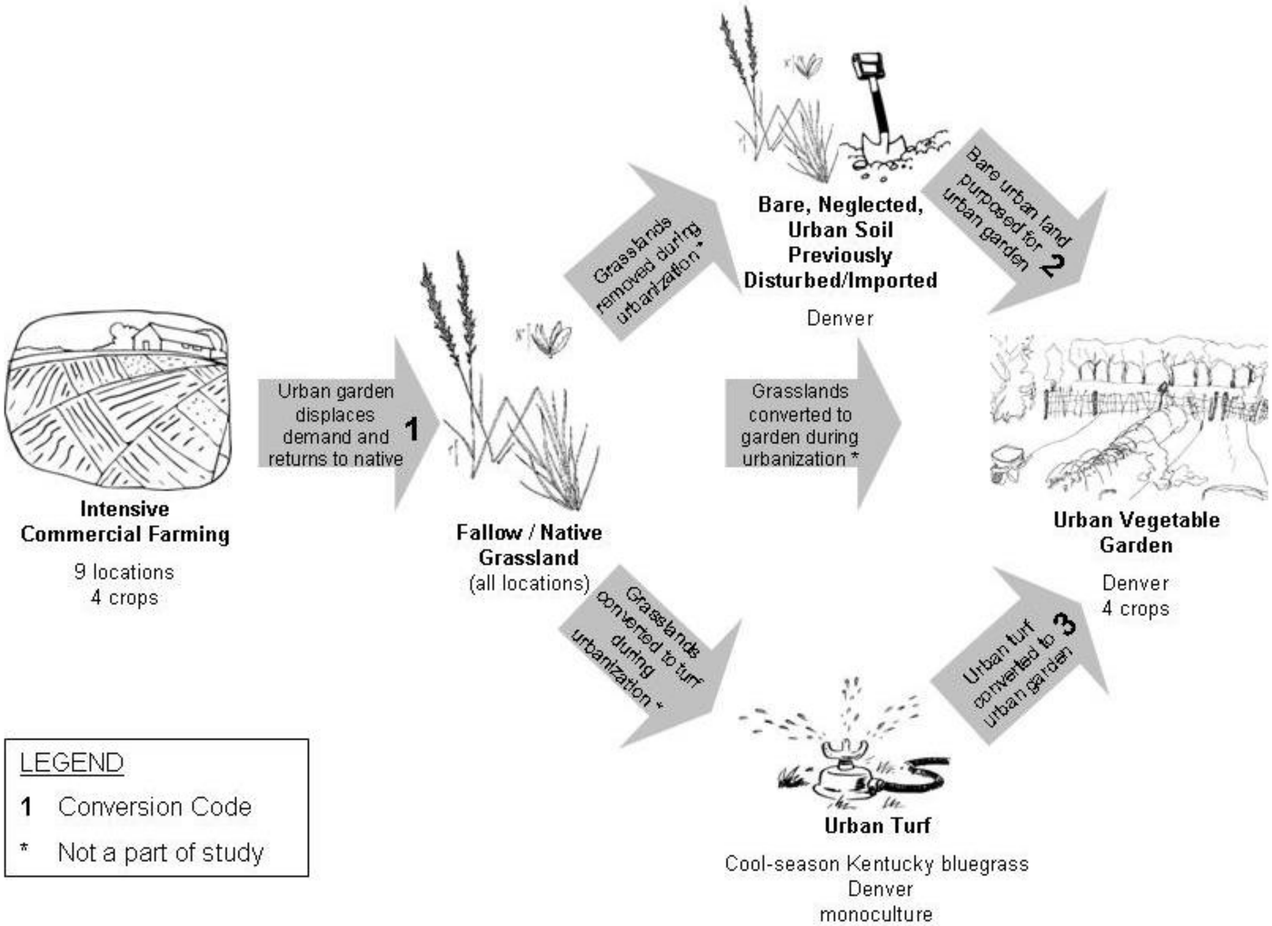

Figure 4. Land Use Change and Conversion Scenarios.

The array and variety of vegetable and production formats combined with land use scenarios results in 19 unique vegetable-land use combinations. In each scenario, a given land use is modeled for 30 years and paired with a subsequent land use for another 30 years. The 30-year time frame was chosen because this amount of time is sufficient to achieve relatively steady-state values for the parameters of interest (SOC, CH_4, CO_2, N_2O emissions), and is a time frame used for environmental analysis. Following the order shown on Figure 4, net changes in impact categories from commercial to urban formats are reported by first calculating the net impacts from two conversions to urban production in the urban setting (from turf production or no production). This is then compared to the net changes in impact categories when commercial production land is put out of production.

3. Results

Life cycle impact categories included direct resource use, upstream resource use, midpoint environmental impacts, labor value, and labor hours. Midpoint impacts were chosen for the life cycle impacts assessment

(LCIA). Examples of midpoint impacts are global warming potential, resource depletion potential, or 50% lethal concentration. Impact methodologies include the Intergovernmental Panel on Climate Change (IPCC) 100-yr global warming potential (IPCC 2007) and the U.S. Environmental Protection Agency Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) which utilizes "the amount of the chemical emission or resource used and the estimated potency of the stressor" (Bare 2011).

The study found that displacement of two previous urban land uses (vacant/degraded and ornamental turf grass) by instances of urban vegetable production had beneficial environmental impacts in terms of greenhouse gas emissions, water use, and soil organic carbon. Displacement of vegetable production on commercial farmland by urban vegetable production also had significant benefits. Most notably, and the basis for application of a social LCA impact methodology, is value of labor in both wages and level of effort. As expected, there is a stark and measurable difference between the labor inputs of large-scale and small-scale production of any kind. This can be seen as an opportunity for local employment, but a risk for corporate efficiency. The data exemplify the significant influence of mechanization and economies of scale for industrial processes. Surprisingly however, urban vegetable production is not categorically favorable for each metric; several key parameters can shift the balance in favor or out of favor for either growing format, and these parameters are distinctly bottom-up. Selected results from the inventory impact assessment are presented in Table 3 and on Figures 5 and 6. Impacts and inventories are not presented for every crop-location combination for brevity.

4. Discussion

The study found that, for some location-crop combinations, smaller resource use and fewer emissions result from urban vegetable production compared to commercial farmland. Displacement of ornamental turf grass in an urban setting by instances of urban gardening had the most beneficial environmental and social impact even when using treated, potable water. Displacement of commercial farm land use by urban garden land use also had significant benefits. Impacts for each crop and conversion scenario were segregated into direct production impacts (direct applied water and energy), indirect production impacts (upstream impacts of products used in on-farm production), and post-production impacts (wholesale packaging and transport to the consumer). For brevity, only two categories (energy inventory and CO_2e impacts) are presented as examples (Figures 5 and 6). In the figures, the stacked categories are direct emissions from production, indirect emissions from production, and post-production emissions. The graphs are arranged by impact and show all crops individually, and the land use change conversion scenarios associated with them. Land use change is presented on a separate y-axis (right hand). The large differences associated with tomato, for example, are owed to the use of lighting to start plants from seed before transplanting, as is common practice for small-scale urban tomato production.

5. Conclusion

Generally, hybrid LCAs are saddled with significant limitations in terms of reconciling scales, data sources, and tracking variability and uncertainty. No quantitative error or distributions were possible due to this approach. The study could have benefitted from a larger sample size of instances of urban production, spanning several years of growing seasons. Because yields affect all impacts linearly, variations in yield can strongly influence whether one growing format is more environmentally friendly than others. Data collection could have been more vigilant and more commitment and communication between the author and growers would have benefitted the amount and accuracy of the primary, case-study data.

These data could allow state and local policy makers to claim, in some instances, that benefits of urban agricultural production outweigh costs. It is clear that multiple independent variables influence impacts; this study offers a first quantitative look at parameters, local practices and conditions that could be further encouraged or modified to improve metrics. The inconsistent relative contributions of different life cycle phases to overall metrics for each crop-location combination were particularly interesting. Another strength of the study is a point of comparison with other LCAs of specialty crop production and market gardening (although few in the literature), shedding light on the importance (or not) of local practices and conditions and potential pitfalls of the adoption of relatively few LCAs applied regionally or nationally. These results can also be used in the larger life cycle assessment and land use analysis of urban areas that recognize many transboundary urban infrastructure contributions, appropriate scales, and their corresponding data sets.

Table 3. Impact/inventory categories by crop and net change due to land use conversions.

	Conversions from Commercial Production to Urban Production							
Category	**Δ2 – Δ1 Potato**	**Δ2 – Δ1 Carrot**	**Δ2 – Δ1 Onion**	**Δ2 – Δ1 Tomato**	**Δ3 – Δ1 Potato**	**Δ3 – Δ1 Carrot**	**Δ3 – Δ1 Onion**	**Δ3 – Δ1 Tomato**
Selected Inventory (unit/lb or as noted)								
Energy (MJ)	-0.429	-1.090	-1.785	9.236	-1.950	-2.612	-3.306	7.715
Land Use (ft^2)	2.493	0.656	-0.195	-0.258	1.479	-0.358	-1.209	-1.272
Water (gal)	7.356	16.355	-10.528	-3.350	-17.807	-8.808	-35.691	-28.514
Soil Organic Carbon (SOC) (Change in kgC/kg soil per year)	-1.32×10^{-04}	-1.55×10^{-05}	-3.57×10^{-04}	-3.03×10^{-04}	-2.55×10^{-04}	-1.39×10^{-04}	-4.80×10^{-04}	-4.27×10^{-04}
Labor Hours* (hours)	0.161	0.055	0.046	0.052	0.155	0.049	0.040	0.046
Labor Pay* (U$S)	$1.28	$0.44	$0.36	$0.41	$1.04	$0.20	$0.12	$0.17
Impacts (unit/lb) [method]								
Global Warming Potential (CO_2e) [IPCC 2007]	-0.007	-0.068	-0.456	-0.121	-0.196	-0.279	-0.666	-0.331
Carbon Dioxide (CO_2 as CO_2e) [IPCC 2007]	-0.016	-0.053	-0.363	0.014	-0.144	-0.208	-0.516	-0.140
Nitrous Oxide (N_2O as CO_2e) [IPCC 2007]	-0.020	-0.028	-0.106	-0.187	-0.027	-0.036	-0.115	-0.195
Methane (CH_4 as CO_2e) [IPCC 2007]	0.005	-0.003	-0.006	0.036	-0.008	-0.011	-0.014	0.028
Human Health Carcinogens (CTUh) [TRACI]	4.33×10^{-09}	2.11×10^{-09}	1.43×10^{-09}	8.08×10^{-10}	-4.57×10^{-09}	$-6.79\times10\text{-}^{09}$	$-7.47\times10\text{-}^{09}$	-8.09×10^{-09}
TRACI Human Health Non-carcinogens (CTUh) [TRACI]	-7.65×10^{-10}	-5.80×10^{-08}	-8.44×10^{-09}	-9.69×10^{-09}	-4.07×10^{-08}	$-9.79\times10\text{-}^{08}$	$-4.84\times10\text{-}^{08}$	-4.96×10^{-08}
TRACI Ecological Releases to Air (CTUe) [TRACI]	0.002	-0.055	-0.012	0.017	-0.001	-0.058	-0.015	0.014
TRACI Ecological Releases to Water (CTUe) [TRACI]	-0.080	-0.257	-0.160	-0.344	-0.382	-0.559	-0.462	-0.646
TRACI Ecological Releases to Soil (CTUe) [TRACI]	-0.021	-0.012	-0.040	-0.132	-0.021	-0.013	-0.040	-0.133

Shaded cells denote reduced resource use or improvements to human health, environment, and society. Unshaded cells denote the opposite.
* The author refrains from interpreting this category as an improvement or detriment.
Inventory and impacts estimated by direct measurement, modeling, or SimaPro Life Cycle Analysis version 7.3.
Values shown based on mean values and confidence intervals (not shown for brevity).
Δ2 – Δ1: Conversions from degraded urban land to urban garden (Δ2) displace demand on large commercial farms. The displacement results in commercial farmland to be left fallow (Δ1).
Δ3 – Δ1: Conversions from urban turf to urban garden (Δ3) displace demand on large commercial farms. The displacement results in commercial farmland to be left fallow (Δ1).
IPCC Intergovernmental Panel on Climate Change 100-yr global warming potential
CTUe Equivalent Comparative Toxicity Units for ecological receptors. Defined as the potentially affected fraction of species (PAF) integrated over time and volume per unit mass.
CTUh Equivalent Comparative Toxicity Units for human health receptors. Defined as the estimated increase in morbidity in the total human population per unit mass of a chemical emitted.
gal Gallons; kg Kilograms; lb Pounds; MJ Megajoules
TRACI U.S. Environmental Protection Agency Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (Bare 2011). http://www.epa.gov/nrmrl/std/sab/traci/

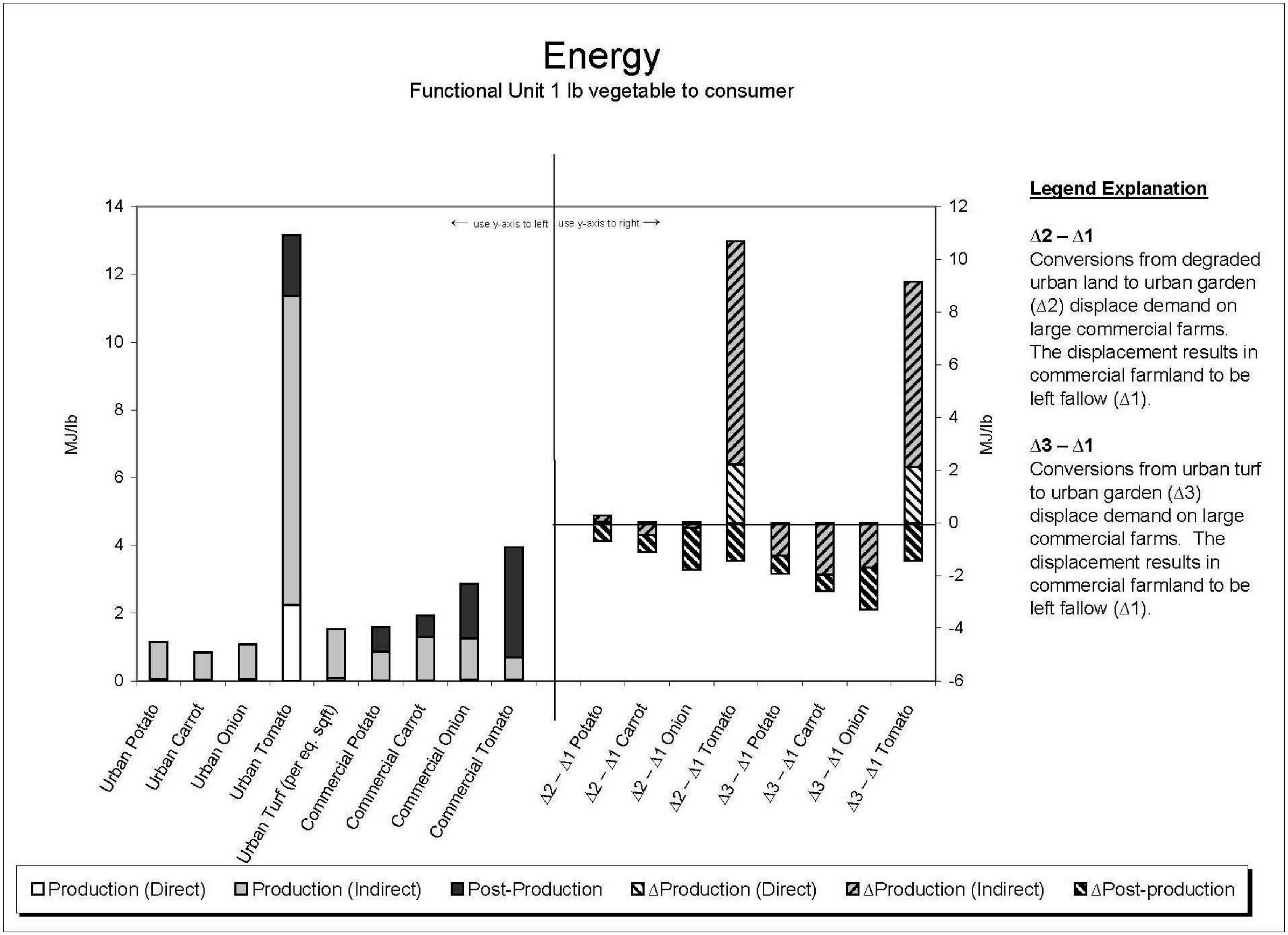

Figure 5. Energy Inventory by LCA Phase.

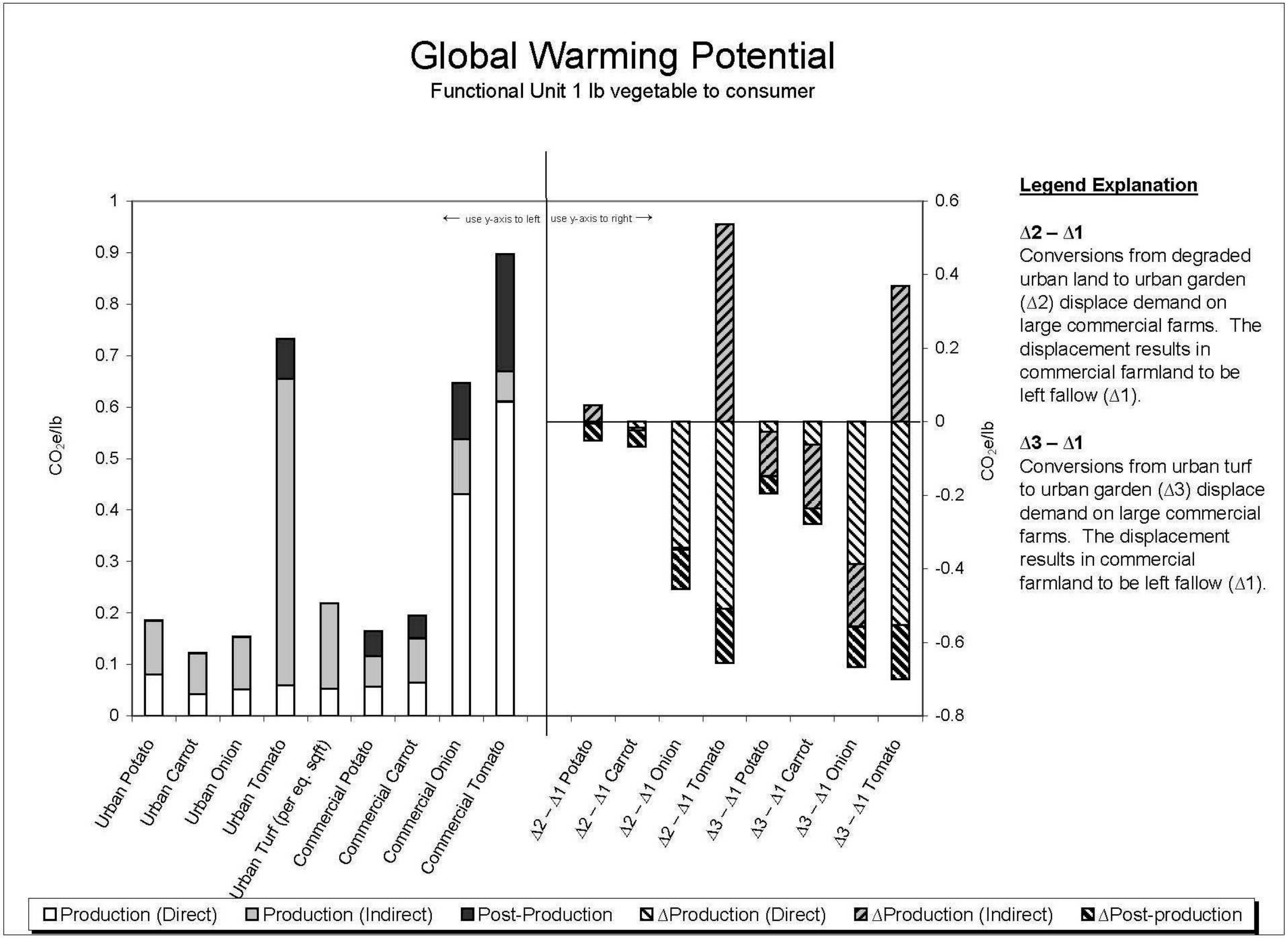

Figure 6. Global Warming Potential Impact by LCA Phase.

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How to use LCA in a company context – the case of a dairy cooperative

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ABSTRACT

The present paper uses the dairy cooperative Arla Foods as an example how life cycle assessment (LCA) can be used within a company. The total carbon footprint for Arla Foods, from cow to consumer, is estimated at almost 18 million tonnes CO_2e, where primary production represents about 80% of the emissions. Arla Foods has defined environmental goals for all stages of the life cycle. The goal for greenhouse gas (GHG) emissions is to reduce emissions with 30% per kg milk at farm level (between 1990 and 2020) and with 25% for processing, transports and packaging (between 2005 and 2020). Another goal is to help consumers to reduce their food waste with 50%. In order to follow up and reach the goals, different tools have been developed to support the environmental work. Some of the challenges faced during the process, especially at farm level, are also discussed.

Keywords: dairy, life cycle assessment, carbon footprint, greenhouse gas emissions, emission reduction

1. Introduction

During recent years there has been a change in the focus of the environmental work among companies. From being a matter of complying with environmental legislations at site level, many companies now also have strategies and targets to reduce the environmental impacts at other parts of the value chain. Life cycle assessment (LCA) is a useful tool to understand the magnitude of environmental impacts at different stages of the life cycle and how these can be reduced. Life cycle thinking is also important to be able to track net-improvements and to avoid (or minimize) shift-of-burden problems, where solutions to one problem becomes the cause of another. The present paper uses the dairy cooperative Arla Foods as an example how LCA can be used within a company context to reduce environmental impact and to become more sustainable. Arla Foods is one of the largest dairy companies owned by about 12000 farmers in seven different countries (Sweden, Denmark, UK, Germany, the Netherlands, Belgium and Luxemburg) and with a global milk intake of almost 13 million tonnes milk.

Arla Foods has worked with reducing the environmental impacts for several decades. Initially the focus was on issues within Arla Foods' own operation, e.g. waste water, emissions to air, noise and odor at site level, followed by reductions in energy and water use as well as logistic planning to reduce fuel consumption. Around year 2000, the focus expanded to include packaging and externally managed transports. In 2008, the concept *'Closer to Nature'* was launched by Arla Foods with environment and climate concerns as an essential element. One of the goals was to reduce greenhouse gas emissions with 25% within the areas processing, transport and packaging between 2005 and 2020. Some years later, in 2011, a new environmental strategy for 2020 was presented, in which Arla Foods promised to take responsibility for the full value chain – 'from cow to consumer'. It was stated that Arla Foods would promote sustainable dairy farming and that specific targets should be developed. At processing level it was decided that 50% of all energy use should be renewable and that there should be zero waste (i.e. all waste should be reused or recycled). Specific goals were defined also at the last stage of the value chain, at consumer level and for waste management, where food waste at consumer level should be halved and all packaging should be 100% recyclable. After a couple of years a dairy farming sustainability strategy was adopted focusing on the areas animals, climate, resources and nature. A climate target was defined at farm level, where greenhouse gas (GHG) emissions should be reduced with 30% per kg of milk between 1990 and 2020. Figure 1 illustrates how the environmental work at Arla Foods has evolved and how Arla Foods has developed specific environmental targets, starting with its own operations (processing, transport and packaging) to finally include the whole value chain. The cradle to gate carbon footprint (CF) for Arla Foods has been calculated previously, but no study of Arla Foods' total CF from cow to consumer has yet been conducted.

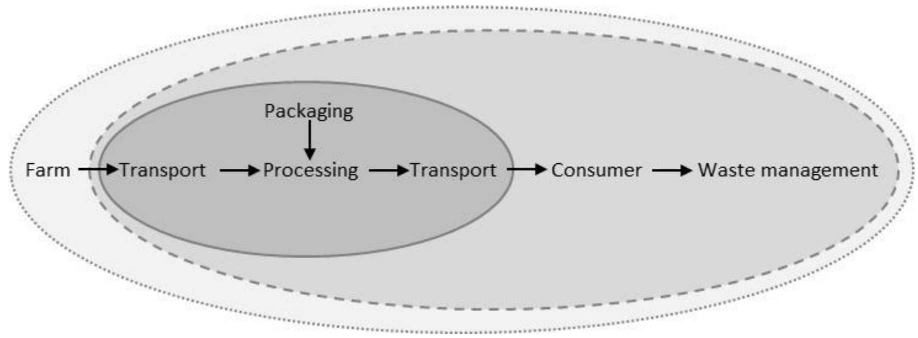

Figure 1. Illustration of how Arla Foods has worked with defining environmental targets within different areas over time, starting with processing, transport and packaging (solid line), then including also consumer and waste management (stippled line) and finally also including farm level (dotted line).

Arla Foods has also established goals on other parameters besides contribution to global warming (such as animal welfare, water and energy). However, the present paper focuses primarily on GHG emissions since this is the area which has gained most attention and where several methodological challenges have been identified.

The overall purpose of the present paper is to show how LCA can be used within a company context in order to reduce environmental impacts. More specifically the total CF for Arla Foods, from cow to consumer, is estimated, as well as the CF for some specific dairy products. In addition some of the main challenges and difficulties faced during the process are discussed. Finally some aspects of Arla Foods future work on sustainability are addressed.

2. Methods

LCA has been the methodological approach for the whole environmental work of Arla Foods, both when setting targets based on the strategies and when defining actions to fulfill the targets as well as following up on targets. The applied methodology related to the different stages of the life cycle is described in the following.

2.1. Farm level

Pre-farm gate emissions constitute the largest share of the environmental burdens in the value chain of dairy products. Raw milk production typically constitutes 80-90% of the CF of milk at the point of distribution to retail (Gerber et al., 2010). It is therefore pivotal to include farm level when promoting a more sustainable dairy production. One critical aspect for the CF result of milk at farm level is which LCA methodology is used: attributional LCA (ALCA) or consequential LCA (CLCA) (Thomassen et al., 2008). The two methodologies obviously answers different questions, where ALCA has a 'what is' focus while CLCA has a 'what if' focus. Some standards leave room for interpretation and leave it up to the practitioner to define the 'rules' for ALCA and CLCA (e.g. ISO 14040 and ISO 14044 (ISO 2006a,b)). Other standards/guidelines give clear guidelines when ALCA and CLCA should be used and how e.g. co-product handling should be conducted in the different cases (e.g. ILCD by the European Commission, Joint Research Institute and Institute for Environment and Sustainability (2010)). There are also some standards/guidelines that specifically recommend a certain LCA method (e.g. PAS2050 (BSI, 2011)). As farm level results in the largest share of GHG emissions of milk, Arla Foods decided to develop a tool that could handle several LCA methodologies in order to answer different questions to assure that reductions are achieved no matter what methodology is used. It resulted in a tool that has a 'switch' that makes it possible to generate results according to CLCA, ALCA (using only economic allocation), PAS2050 (Carbon Trust, 2010) and IDF (IDF, 2010) respectively, using the same input data (Dalgaard et al., 2014).

However, it is not feasible to report and communicate results from four different methodologies. Arla Foods has therefore decided to follow the methodology from the International Dairy Federation (IDF, 2010), developed by the global dairy industry, to report and deliver on the goal of reducing GHG emission. Since there is no agreed method on how to deal with emissions from land use change (LUC) Arla Foods do not include LUC emissions as default.

In the calculations of the total CF of Arla Foods in the present paper, the average CF of milk at the farm gate is estimated to be one kg CO_2e per kg milk, as no more detailed and uniformly calculated CF data are available in all countries where Arla Foods has production. This estimate is based on reviewing literature (Flysjö et al., 2011a; Hagemann et al., 2011; Kristensen et al., 2011; Williams et al., 2006) and comparing with the CF for milk in Sweden and Denmark in 1990 and 2005, and in Germany and UK in 1990 (Figure 3) (Dalgaard and Schmidt, 2012; De Rosa et al., 2013; Schmidt and Dalgaard, 2012), and is considered realistic and even somewhat conservative. As described above, emissions associated with LUC is not shown in the results in the present paper, however, the tool developed for calculating the CF of milk at farm level has a 'switch' to include missions from LUC using different methods. The different features of the CF tool are further addressed in the discussion.

2.2. Processing, transport and packaging

Arla Foods is collecting data on a yearly basis for all energy use for and waste from processing, transports (own operations as well as externally managed) and packaging. The data is used to follow up on reduction goals and is used in the present study to calculate the total CF of Arla Foods. Dairy products such as cheese, butter, milk powder and whey are also purchased together with 'other' raw materials and ingredients such as jam, sugar, salt and vegetable oil. To estimate the CF for purchased dairy products the method presented in Flysjö *et al.* (2014) is used, and data for other raw materials and ingredients are obtained from Davis et al (2011), Ecoinvent (2010), Schmidt (2007) and SIK (2009).

In the present study, the CF for some general dairy products is presented; whole milk (3% fat), yoghurt (3% fat), butter blend (with 60% fat of which 63% butter fat and 37% vegetable oil) and yellow cheese (17% fat). The packaging for the different products are one liter paper carton with plastic cap for milk and yoghurt, 250 gram tub for butter blend and plastic foil for cheese (packaging size of 800 gram). One of the most critical decisions for the CF of dairy products is co-product handling. To calculate the CF for the products mentioned above the method in Flysjö et al. (2014) is used, where allocation of raw milk is done based on the value of the different milk solids (fat, protein and lactose) in the final products.

2.3. Consumer level

To analyze the CF of the whole value chain of Arla Foods, the consumer stage also needs to be accounted for. In the present paper data for GHG emissions from retail, home transport and storage in refrigerator at consumer have been added to the total emissions of Arla Foods. Data for the latter part of the life cycle is taken from Flysjö (2011) and Flysjö (2012). Another aspect that can impact the CF value is food waste, especially at consumer level. This is also analyzed and estimated numbers on food waste at consumer used in the present study are 4% for milk, 10% for yoghurt, 3% for cheese (Berlin et al., 2008) and 5% for butter blend (Flysjö, 2011).

3. Results

The total CF – from cow to consumer – of Arla Foods in 2013 is estimated to almost18 million tonnes CO_2e (Figure 2). Milk and raw materials stand for the largest share of the CF (about 80%), while processing, transport and packaging stand for about the same amount of emissions as consumer stage. Emissions associated with raw milk production represent the largest share (almost 90%) of the first bar in Figure 2, whereas purchased dairy products stand for about 10% and other raw materials and ingredients stand for less than 2%. The largest share of direct emissions at consumer stage is from home transport (about 70%). Food waste is not shown in Figure 2 as these emissions are related to up-stream activities. Overall, however, food waste is likely to represent the largest environmental impact at the consumer stage.

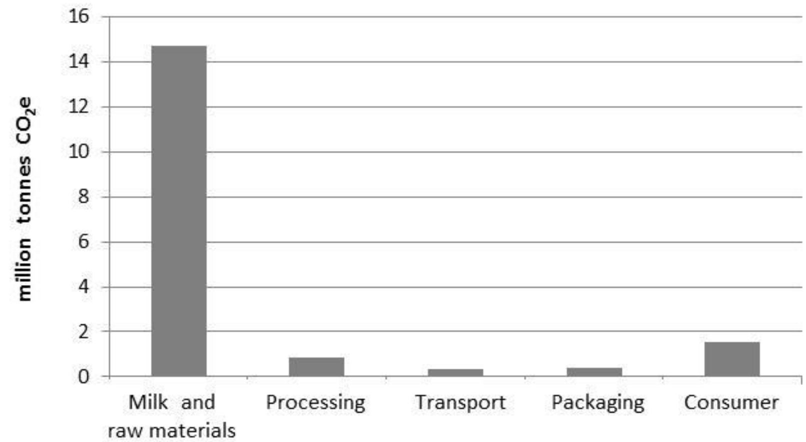

Figure 2. Total CF of Arla Foods production, from cow to consumer.

The CF per kg of fat and protein corrected milk (FPCM) at farm gate was about 1.25-1.3 kg CO2e in Sweden, Denmark, Germany and UK in 1990 (using the IDF methodology and excluding emissions from LUC) (Dalgaard and Schmidt, 2012; De Rosa et al., 2013; Schmidt and Dalgaard, 2012). In 2005 the CF per kg FPCM was reduced by 15% and 22% for Sweden and Denmark, respectively. No CF results have been estimated for Germany and UK for 2005. The stippled line in Figure 3 shows the reduction goal of -30% emissions per kg milk in 2020, compared to 1990.

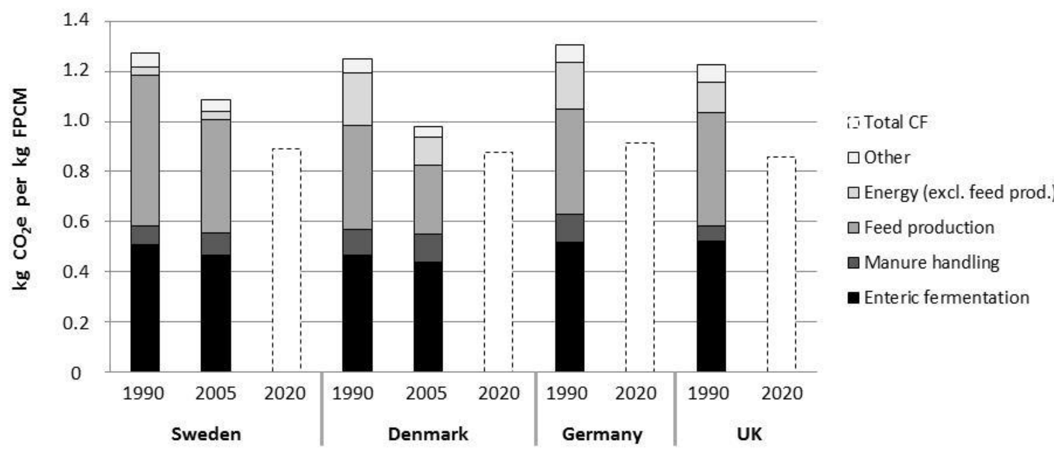

Figure 3. The carbon footprint of fat and protein corrected milk (FPCM) at farm gate in Sweden, Denmark, Germany and UK in 1990 and 2005 (latter only for Sweden and Denmark). Stippled line shows 30% lower CF compared to 1990, which is Arla Foods target at farm level.

Figure 4 shows the total reductions for Arla Foods in GHG emissions for processing, transport and packaging from 2005 to 2013. Most reductions have been achieved for processing and packaging (about -15%, respectively), while transport is about same in 2013 as in 2005. There are uncertainties in the numbers, as Arla Foods has been through a number of mergers and acquisitions since 2005 and reporting might not be the same in other companies.

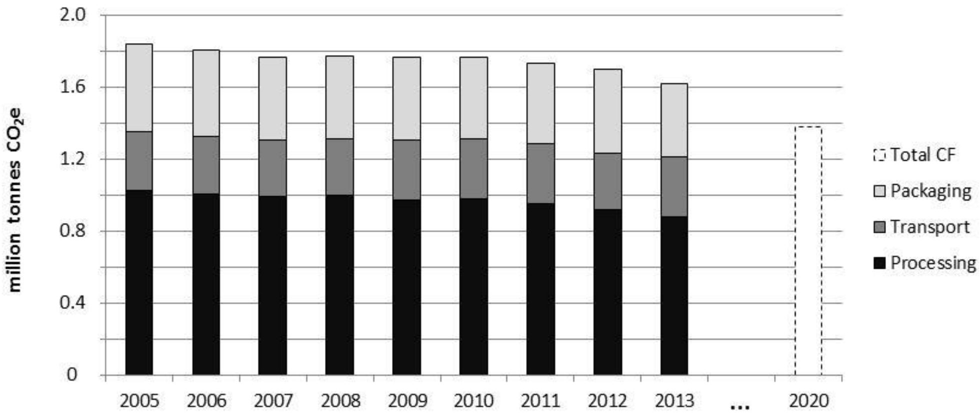

Figure 4. Arla Foods' total reduction in greenhouse gas emissions from processing, transport and packaging from 2005 to 2013. The stippled line shows the goal of -25% emissions in 2020.

Figure 5 shows the relative distribution of GHG emissions between the different life cycle stages for milk, yoghurt, low fat butter blend and yellow cheese, both including and excluding waste at consumer level. The estimated CF for each product including waste at consumer is 1.4 kg CO_2e per kg milk, 1.6 kg CO_2e per kg yoghurt, 7.1 kg CO_2e per kg butter blend and 8.8 kg CO_2e per kg cheese. Retail and consumer level emissions constitute a relatively larger share of the CF for milk and yoghurt, as these products has a lower CF per kg product compared to butter blend and cheese.

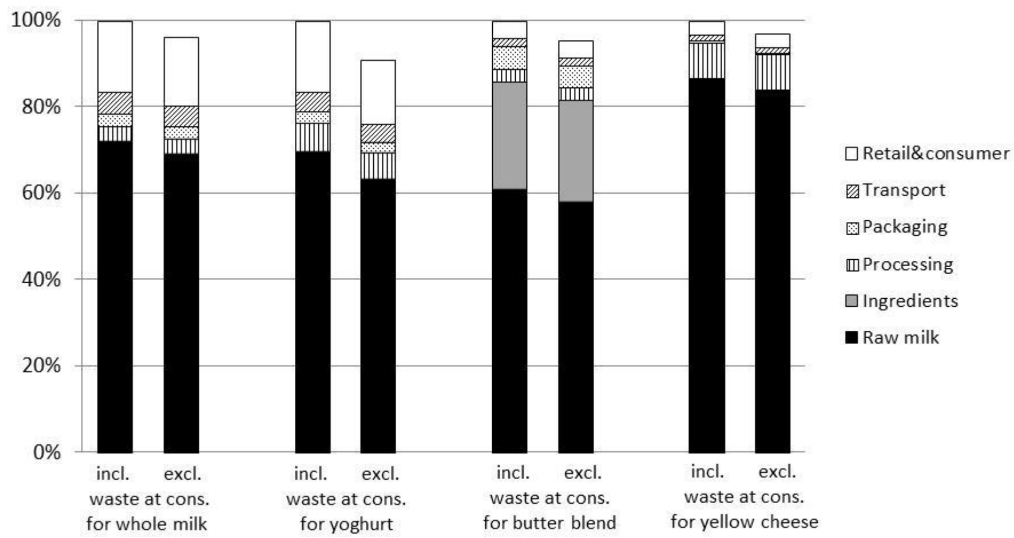

Figure 5. Relative distribution of the carbon footprint between life cycle stages for the different dairy products; milk, yoghurt, butter blend, and yellow cheese, including and excluding waste at consumer level. Ingredients in butter blend is vegetable oil.

4. Discussion

Having a life cycle perspective is important to achieve efficient environmental improvements and to ensure net benefits throughout the value chain. Initially, focus was put on Arla Foods' own operations, but today the whole value chain is included. Figure 6 shows the relative environmental impact of the different life cycle stages and illustrates the possibility Arla Foods has to influence the different life cycle stages. Farm level is obviously the life cycle stage which stands for the largest share of emissions for dairy products, followed by the consumer stage (Figure 2). Thus, from a life cycle perspective, farm level and consumer level would be the two life cycle stages to focus to reduce emissions most efficiently. However, these two life cycle stages are also the ones most challenging to influence. Some of the challenges related to the LCA methodology and data gathering are discussed below. One of the critical aspects is to find a model that captures enough detail and relevant aspects of GHG emissions at farm level to ensure that the goals are reached, i.e. that actual reductions are achieved.

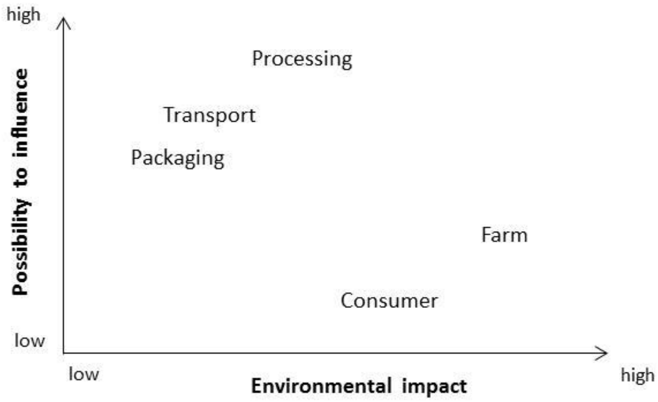

Figure 6. Illustration of which life cycle stages have the largest emissions and where Arla Foods has the largest influence.

4.1. Farm level

One of the main differences between primary production and the rest of the value chain is that methane (CH_4) and nitrous oxide (N_2O) constitute the largest part (70-80%) of the GHG emissions, while for processing, transport, packaging and consumer level it is carbon dioxide (CO_2) from combustion of fossil fuel that is the main contributor to the CF. There is a relatively large certainty in the CO_2 emissions from combustion of fossil fuel and a reduction in these emissions is thereby relatively certain. When it comes to emissions of CH_4 and N_2O on the other hand, there is a larger inherent uncertainty in these emissions (Flysjö et al., 2011b). The main share of the CH_4 is from enteric fermentation when the cow is digesting the feed and is a naturally process for ruminants in order to make use of grass and other feed sources inedible for humans. The largest share of N_2O emissions is from nitrogen turnover (from synthetic fertilizers, manure/excreta and crop residues) in agricultural soils used for feed production and is also a natural and inherent variable process. Hence, there will always be CH_4 and N_2O emissions from milk production due to the characteristic of milk and agricultural production. Also, the CF of milk at farm level is related with a relatively large uncertainty, which obviously makes it more difficult to estimate total emissions and to follow up on reduction measures. In some cases reduction measures can be known (e.g. drainage of soils), but not captured in the models/calculations and thereby not part of the reported emission reductions as it is not possible to quantify them. Another challenge is the representativeness. Arla Foods has over 12000 dairy farms in seven different countries delivering milk, and it is not realistic to collect data from all these farms. The ease at which a CF for an individual farm can be calculated is very much related to data availability and differs between countries. In Denmark farmers report more data to statistics compared to e.g. Sweden and Germany. Hence, in Denmark it would be easier to calculate the CF for a larger number of farms compared to the other two countries. However, Arla Foods want to have a similar approach in all countries and plan to calculate the CF for about 200 farms yearly in Sweden, Denmark, UK and Germany, respectively (Germany also include the farmers in the Netherlands, Belgium and Luxembourg). The number of CF assessments at farms is assumed to be more than enough to be representative according to the Carbon Trust (2010). Today about 1500 CF assessments have taken place together with almost 300 farm workshops that Arla Foods has arranged with focus on sustainable dairy farming in Sweden, Denmark and UK. However, it needs to be evaluated whether it is possible to use these ‘bottom-up’ CF numbers to follow up on the overall reduction goal, where the baseline for 1990 was calculated ‘top-down’ using national statistics. It is important to keep in mind that CF numbers should not be compared unless they are estimated in a comparable way (Henriksson, 2014). At the moment different support is also provided from authorities in the different countries regarding calculating CF at farm level and extension services coupled to these calculations. It is important to make use of national initiatives, but risks introducing differences in methodology. The feature with the Arla tool was to use the same tool, but with country specific background data, to calculate the CF for milk for different countries. However, due to data availability and data transfer this is not up and running for the moment. Another feature with the Arla tool was to be able to use the same input data and generate results according to different LCA methodologies. Being able to present CF results using different methodologies can be useful in some situations. While the IDF methodology is chosen for reporting and in communication with farms, the CLCA approach can be useful on a policy level and to understand e.g. the link between milk and meat production and what effect milk production has on other systems. Having the different methodologies in the tool can also be used to assure that improvements are taking place. This was shown when calculating the CF for milk in Sweden and Denmark in 1990 and 2005, where there had been a reduction in emissions between the years no matter which LCA methodology was used (results not shown).

4.2. Processing, transport and packaging

For the next part of the value chain, on the other hand, it is much easier to ensure that reductions are achieved, since a reduction in e.g. fuel oil results in lowered emissions. On-site energy use is also within the direct control of Arla Foods and it is thereby easier to implement actions to ensure reductions. The same also goes for own operation transports. In many cases, however, infrastructure can be limiting and it can thereby be difficult to e.g. increase the use of renewable energy sources. For the externally managed transport it is important to work with suppliers and request transports with lower emissions. When it comes to packaging there are several aspects to consider. Different materials have different environmental impact, so the choice of the material is obviously important. The packaging should preferably also be possible to recycle, why also the end-of-life needs to be considered. Again, a limiting factor can be infrastructure, such as waste management systems in the country where the products are sold, since not all countries have a recycling system for waste. To put focus on the environmental impact of packaging, Arla Foods has developed a tool to analyze the CF of different packaging solutions (materials, end-of-life solutions etc). This allows incorporating environmental thinking already in the design of the packaging. The function of the packaging is another aspect that needs to be considered. The trend today is that there are more single households, more advanced packaging (e.g. milk cartons with plastic cap, sliced cheese in plastic pack) and more 'on the go' meals (one portions with plastic spoon etc). All this results in more packaging per kg of product, but at the same time such packaging can contribute to less food waste. So in total there might be a net benefit considering the whole life cycle of the packaging (including the product).

Arla Foods are reporting its total GHG emissions according to the greenhouse gas protocol (World Business Council and World Resource Institute for Sustainable Development) and the goal on reducing the GHG emissions is established according to that. The 25% reduction is in absolute numbers, i.e. not per kg of product. However, if the company is expanding through mergers or acquisitions the baseline needs to be recalculated, but any organic growth is not accounted for. Since the reduction goal does not account for the amount of products produced, it would also be desirable to calculate the CF per product or product group, to also capture the efficiency of production. Today Arla Foods has developed a method to calculate the CF of dairy products (Flysjö et al., 2014) and has assessed the CF for a number of dairy products (Flysjö, 2012). The next step is to implement the model on site level to follow improvements on product level. This would for example account for reductions in product losses which currently are not reflected in the reporting according to the GHG protocol.

4.3. Consumer level

Home transport showed to be the activity leading to the main share of direct emissions at consumer stage (Figure 2). This estimate is obviously related with rather high uncertainty, and little information is found on consumer travels associated with food purchasing. Sonesson et al. (2005) suggests 28-63 km per week and household to be a reasonable assumption for food purchases in Sweden. The present study uses the lower value when estimating Arla Foods total CF.

Another aspect, probably even more important to focus on at consumer level, is food waste. Even though the emissions associated with food waste occurs up-stream of the value chain (e.g. at farm level, processing, transport), the consumer has the possibility to reduce the amount of food waste and therewith reduce the amount of food that is produced for no reason. A study initiated by the Food and Agricultural Organization of the United Nation (FAO) estimated that 7% of dairy products are wasted at consumer level in Europe and in US the number is about twice as high (Gustavsson et al., 2011). Sonesson et al. (2005) found even larger waste numbers for dairy. Assuming the food waste of dairy products to be 7% it would equal a CF on 1.3 million tonnes CO_2e, and if the waste would be 15% it would be about 2.7 million tonnes CO_2e, for Arla Foods total production.

Arla Foods has a goal on reducing food waste at consumer with 50% by 2020 and are analyzing ways to help and inspire consumer to avoid wastage of food. Among other things an 'empty your fridge' app for smart phones is developed (for the Danish market), with recipes to help consumers make use of left overs. Packaging design is another way to reduce food waste and is one aspect Arla Foods is working with. A study analyzing the CF of butter showed that 37% of the butter in mini-tubs was wasted (Flysjö 2011). Arla Foods has now redesigned the packaging which now consists of 8 gram butter instead of 10 gram. There has not yet been any study to follow up the results, but less butter in mini tubs are likely to be wasted.

4.4. From minimizing negative impact to maximizing positive impact

During the last decades, the environmental work of Arla Foods has shifted from focusing on single impacts (e.g. emissions and resources) at only part of the value chain where Arla Foods has direct influence (e.g. processing), to include several impact categories to better capture the whole concept of sustainability (e.g. GHG emissions, resources, animal welfare, biodiversity) for the entire value chain from cow to consumer (Figure 7). Hence, the scope has broadened both regarding focus areas/impact categories as well as stakeholders/value chain.

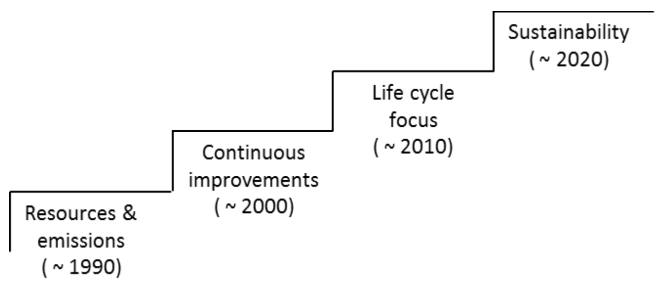

Figure 7. The development of focus areas for Arla Foods' environmental work (inspired by Nielsen and Remmen, 2005).

One recent example is that in order to further improve the sustainability of the production, Arla Foods has decided to purchase RTRS certified soy (for all soy fed to the dairy cows delivering milk to Arla), certified palm oil (used in dairy products) and certified cacao (used in dairy products). There is also some other 'sustainability' projects on e.g. tree planting in Mozambique and Uganda.

As a cooperative, Arla Foods has a close dialogue with the dairy farms. This creates a unique possibility to work together to improve and reduce GHG emissions at farm level and promote a more sustainable dairy production. At the same time, however, it can be challenging as the farmers are the owners of Arla Foods and it is therefore not possible to simply define targets and put demands at farm level without their accept. In order to influence farmers a combination of awareness raising and motivation has to be adopted. Similarly, in order to achieve the goals related to consumer food waste, focus needs to be on motivating consumers to make more sustainable choices in addition to offering them e.g. packaging sizes that will enable them to avoid throwing valuable food away. Arla Foods has a long tradition of communicating closely with the consumers, through recipes, home pages, social media and apps for smart phones. The "empty the fridge" app for smart phones, mentioned earlier, is one concrete example on this. Other activities are advices on how to store food, recommendations on portion sizes and advices for "food planning".

The focus of Arla Foods so far is mainly on minimizing the negative environmental impact – the footprint. However, it is also important to focus on maximizing the positive impact – the handprint. Handprints can be generated by reducing the footprint, helping someone else reducing their footprint or taking other generative action (e.g. tree planting) (G Norris, Harvard School of Public Health, Boston, Massachusetts, USA, personal communication). By broadened the scope to include e.g. consumer stage, Arla Foods is trying to inspire others to reduce their footprint. Actively engaging with consumers and other stakeholders, and creating a dialog on sustainability, is more about generating a positive impact and creating a handprint than just reducing the footprint.

Thus, shifting from only focusing on minimizing the environmental impact to maximizing the positive impact is important for a sustainable development.

5. Conclusion

During the last decades, the environmental work of Arla Foods has grown in scope and includes now the full value chain. The broadened scope has also resulted in more challenges. LCA is a valuable tool to report and focus the environmental work. The total CF of Arla Foods from cow to consumer is estimated at almost 18 million tonnes CO_2e. About 80% of the emissions occur at primary production while consumer stage represents about 10% of the CF. Environmental goals are defined for all stages of the life cycle, including consumer level. Between 1990 and 2005 GHG emissions at farm level have been reduced with about 15-20% in Sweden and Denmark. The average reduction in GHG emissions for processing, transport and packaging is 12% between 2005 and 2013. One of the goals at Arla Foods is to help consumers to reduce food waste with 50%. If this goal is achieved it would save about 0.6 million tonnes CO_2e of Arla Foods CF (assuming that 7% of all dairy products are wasted, as estimated for Europe).

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Life Cycle Assessment towards a Sustainable Food Supply – A review of BASF's Strategy

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ABSTRACT

Life Cycle Assessment (LCA) and approaches based thereon, e.g., Eco-Efficiency Analysis and SEEBALANCE, have proven useful tools for a quantitative sustainability assessment along value chains and across industry sectors. In order to holistically assess sustainability in agriculture, AgBalance™ combines LCA with environmental, economic and social impact indicators, generalized to varying spatial scales. The methodology comprises up to 70 sustainability indicators, which are grouped into the three dimensions "environment", "society" and "economy". Sensitivity and scenario analyses can be employed to study the robustness of the model results, and to investigate trade-offs or the response to external influences. In order to better translate findings from the socio-economic LCA studies to continuous improvement on farm, web-based crop management applications are developed. They interactively conduct scenario analysis for a set of sustainability indicators that were shown to be the most relevant ones in an AgBalance case study. This new strategy is designated "AgBalance Farm".

Keywords: Socio-economic LCA, agriculture, AgBalance, continuous improvement, AgBalance Farm

1. Introduction

1.1. Sustainability in the Food Chain

A focus on sustainability in the food value chain has become a basic prerequisite for suppliers and consumers. There is no other industry where so many product and production characteristics are marketed as "sustainable" as in food production (Frank et al. 2013). If sustainability is manifested and proven with different methods, varies widely in the different value chains of the food sector. Sustainability is mainly driven by the farming activities of the farmers who continuously implement new practices and improvements, in a challenging environment of weather, market needs and consumer expectations. Numerous existing sustainability programs often have their roots in Corporate Social Responsibility (CSR) activities, which are not integrated into the core business of the company (Porter and Kramer 2011). However, more and more companies recognize that these types of initiatives not only represent an investment in their brand and reputation, but they can also help to effectively resolve of one of their primary problem areas: supply chain security. Lawrence and Burch (2007) report that food producers and supermarkets increasingly rely on cooperation with suppliers and long term contacts for a secure supply of high quality products, particularly in product categories with high potential for differentiation. These contacts may include multiple layers of the value chain concerned, as they strive for considerably closer cooperation with the producers (Nestlé 2013).

1.2. Sustainability Assessment in BASF

Against this background, there is a demand to be able to map the whole food value chain with respect to the contribution of each stage to the sustainability footprint of the entire chain. BASF comprises one of the few companies being active on all different stages of the food value chain, starting from seed provision, plant protection and fertilizer additive manufacturing, provision of food ingredients, packaging and many more. Therefore, BASF has pioneered an array of LCA tools to capture the impacts on the various levels of the value chain. Tools such as Eco-Efficiency Analysis (Saling et al. 2002) and SEEBALANCE (Kölsch et al. 2008) are used by BASF and its customers to assist strategic decision-making, facilitate the identification of product and process improvements, enhance product differentiation as well as to support the dialogue with opinion makers, NGOs and politicians. Both Eco-Efficiency Analysis and SEEBALANCE analysis are comparative methods; the advantages and disadvantages of several alternatives are assessed according to a predefined customer benefit with a holistic approach. The analysis uses a Life Cycle Assessment approach with the whole life cycle of a product – from cradle to grave – being considered. Next to the environmental impact, which is assessed based on ISO14040, ISO14044 and ISO 14045 norms, all economic factors are taken into account. The SEEBALANCE also considers social impacts of products and processes (Uhlman & Saling, 2010).

Both Eco-Efficiency-Analysis and SEEBALANCE have been employed in the food and feed value chains in order to assess the key drivers of sustainability in various production systems. It has been shown in various case studies that agriculture can have a large share of the entire sustainability profile of food and feed value chains. At the same time, logistics, transport, processing and, not least, consumption can play a substantial role as well. In 2012, for example, BASF analyzed the CO_2 balance for veal and beef products with the client Westfleisch, supporting them in improving the sustainability of their meat production along the whole value chain (Westfleisch 2010.).

Feed ingredients for a more sustainable aquaculture of salmon have been identified through Eco-efficiency Analysis in the collaboration with Biomar A/S (Saling et al. 2007). Three ways to produce astaxanthin as an ingredient of salmon diets were compared: chemical synthesis, fermentation and production via fermentation of algae. In this case study, the astaxanthin derived from chemical synthesis was the most eco-efficient product (Saling et al. 2007). Other examples comprise the production of beef with Cattlemen's Beef Board and National Cattlemen's Beef Association (NCBA 2014). The Eco-Efficiency Analysis portfolio shows that the present-day U.S. beef value chain is more sustainable than in 2005. While there was detected a 6 percent increase in the price of beef between 2005 and 2011, there was a simultaneous decrease in the overall environmental and social impacts from the U.S. beef value chain of approximately 7 percent.

Case studies were also performed on food packaging. In collaboration with the German dairy company Müllermilch, for instance, plastic cups made of polymers turned out to be the more sustainable option compared to returnable glass containers and composite cartons (BASF 2003). In a similar approach, various bottling alternatives for carbonated mineral water were compared. The customer benefit involved consumption of 1000 l mineral water at a distance of 300 km from the bottling plant. The disposable 'Office Line', a novel packaging method for office use, option outperformed the disposable PET or carton as well as the reusable glass bottle due to the favorable environmental footprint (Gerolsteiner 2005). The main sources of impacts are the material production, the cleaning steps and the recycling options for the different alternatives. Transportation plays a minor role. The relation of volume and packaging material and the reuse-cycles is very important for the position in the sustainability portfolio. The limitation of Eco-efficiency and SEEBALANCE to cover the agricultural production level is the lack of specific indicators, among others capturing the impacts on biodiversity, soil health and the agri-sociological context of production. For this reason, BASF has developed AgBalance™: a holistic method for assessing sustainability in agriculture and identifying key drivers for improvement.

2. AgBalance – Life-Cycle Assessment in Agriculture

AgBalance comprises a multi-criteria life cycle based approach in combination with a defined aggregation and summary of single results into a single sustainability score (Frank et al. 2012). AgBalance™ delivers results that enable farmers, the food industry, politicians and society to objectively evaluate processes in terms of their sustainability profile. In doing so, a vast amount of information on individual factors can be ascertained in addition to overall statements on the sustainability of agricultural practices (e. g. ploughing). AgBalance was finalized in mid of 2011. In September 2011, the methodology was given independent assurance by the global expert agencies TÜV SÜD, DNV Business Assurance and NSF International. AgBalance™ can be used to map an individual farm or the whole agricultural sector in one region, for example. The focus can either be on the agricultural production system alone or on the processes that have established themselves downstream in the value chain, such as logistics or processing.

A case study with the holding company SLC Agricola in Brazil involved an internal benchmarking of two large farms, each with over 10,000 hectares, to identify the central sustainability drivers for their crop rotation consisting of soya, maize and cotton and to derive follow-up opportunities for their continuous improvement. An average cultivated hectare for each of the two farms, Panorama (Bahia state) and Planalto (Mato Grosso do Sul state) were compared on the basis of the operation data from the 2009/2010 season. The indicators from all three sustainability dimensions – environment, economy and society – were investigated using a holistic approach over a section of the life-cycle that starts with the raw materials used in the production (the "cradle" of the process, for example phosphorus extraction or oil production) and ends with the delivery of the harvested goods at the nearest port. The analysis revealed that the Planalto farm is substantially more sustainable than the Panorama farm (Fig 1), which is largely due to better results in the economy and environment dimensions.

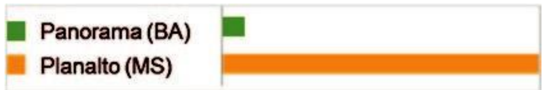

Figure.1. Relative sustainability index of the two farms Panorama and Planalto. Planalto achieved a 40% better result

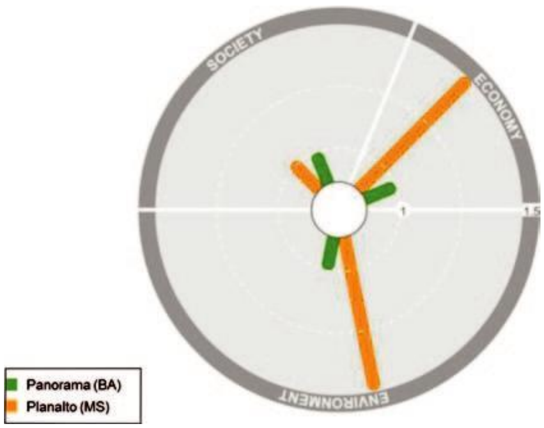

Figure 2. Representation of the sustainability index in terms of the three dimensions of sustainability. The length of the bar indicates higher sustainability. Each time the worse alternative is normalised to the value 1

Whereas the result in the social indicators did not exhibit significant differences (Fig. 2), the most important drivers in terms of economy turned out to be an improved cost situation and an increased profit of the Planalto farm. In terms of the environment, the key drivers turned out to be a predicted imbalance for nitrogen and above all, phosphorous in the soil as well the pesticide regime. According to initial calculations, the optimization of the fertilization regime in Panorama could lead to savings of almost 15 million kWh of energy (this corresponds to the energy use of roughly 2,000 households in Brazil) in addition to substantial cost savings. The CO2 equivalents saved using AgBalance™ amount to almost 8,000 tons per year. These results, together with the additional findings on pesticides, can serve as the starting point for a continuous improvement program at SLC Agricola. With its knowledge base, BASF supports a suitable product portfolio throughout the whole life-cycle and works towards creating common solutions towards greater sustainability.

Measuring sustainability can be a central key to steady improvements towards sustainable agriculture. It is therefore an essential requirement that it succeeds in translating results from complicated life-cycle analyses into farmers' everyday reality and to derive specific recommendations for action. However, agricultural production globally is strongly dependent upon smallholder farming, which is not easily accessible by complex and expert-based LCA approaches. Novel IT solutions are required in order to make use of LCA-based knowledge for a more sustainable crop management on-farm. This is the basic idea of the concept "AgBalance Farm".

3. AgBalance Farm - From LCA to Farm Management

Global food security and sustainability stands and falls with the adoption of sustainable crop management practices by millions of smallholders in developing countries. As outlined above, Life Cycle Assessment has proved capable to reveal the key drivers of sustainable agriculture and thus to serve as a guardrail for improvement strategies (Frank et al. 2012). However, the translation of the results of LCA studies into on-farm decision support has mostly failed. Here, we present a strategy designated "AgBalance Farm" that uses key learnings of Socio-Economic LCA studies for the development web-based crop management support applications for farmers. This strategy is based upon BASF's experience with the Eco-efficiency Analysis manager as outlined in Saling (2013). India is the fifth largest producer of soybean in the world but soybean yields currently reach only half the global average of 2.4mt/ha. Lack of knowledge about good farming practices comprises the key reasons for the low productivity. Through the training program 'Samruddhi' (Sanskrit for 'prosperity'), farmers are educated not only on the timely usage of crop protection inputs, but also about correct fertilization, seed rate and spacing to enable higher yields (GIZ 2013). A group of self-reliant farmers, the so-called 'Margdarshaks', are trained by BASF's technical advisors and are entrusted to promote 'Samruddhi' in their village and to help other these farmers to adopt the best practices. While the contribution of Samruddhi to the profitability of the Indian soybean farmers had been shown (PWC 2013), its contribution to the sustainability of the production was largely unknown (Voeste 2012). Against this background, a holistic socio-economic life cycle assessment using AgBalance™ methodology was conducted. As a test case, soybean production under 'Samruddhi' and 'non-Samruddhi' in the state of Madhya Pradesh were compared. The AgBalance™ revealed that the 'Samruddhi' production practice outperfomed 'non-Samruddhi' in all three dimensions of sustainability. In the economic dimension, a better cost position (fixed & variable) and higher profits per ton of soybean resulted in a better score. In the social dimension, a stronger emphasis on professional training favoured the 'Samruddhi' practice. In the environmental dimension, the better performance of the 'Samruddhi' practice in some LCA impact categories was accompanied by the inferior scores in categories such as soil health, biodiversity potential and emissions. As the key driver for this, the fertilizer regime of 'Samruddhi' was identified (PWC 2013). Based upon this AgBalance™ study, 12 sustainability indicators with a big impact on the study result were selected. Through regression analysis of data sets of approx. 100 individual farmers, mathematical functions describing the interdependencies between the respective indicators were derived, e.g. between yield and nutrient management. A web-based application was generated that can be used e.g. by 'Margdarshaks' to help soybean farmers in their villages optimizing their production protocol towards higher yield, profitability and sustainability. It basically conducts scenario analysis interactively, as demonstrated for the concept of the Eco-efficiency Analysis manager. With this "AgBalance Farm" strategy, we aim to effectively use the potential of socio-economic LCA to support crop management decisions of individual farmers.

4. Summary

Sustainability is becoming increasingly important as a key factor for growth and value creation. Customers along supply chains want more sustainable products and system solutions. There is a need to integrate sustainability much more closely into businesses and decision-making processes. To manage this in an effective way and to support decision-making processes, a sustainability evaluation toolbox is needed which can be applied to assess products and processes in a holistic manner. Both detailed in-depth results of individual impact indicators, as well as aggregated results and a single sustainability evaluation score are output of the sustainability Evaluation methods. Different types of footprinting in combination with other information can support decision-making efficiently. The communication of the results is a key aspect of this type of studies, especially in the food supply chains, where also end consumers are involved. Holistic, LCA-based and scientific sound methods to measure sustainability are key success factors for the realization of more sustainable production systems. Nutrition of human beings will be one of the key challenges of the future and can only be realized in a more sustainable manner. BASF cooperates with all players and producers in the supply chain, from basic production and farming via processing, packaging, transportation and preparation of foodstuffs.

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The link between LCA and CSR with espresso coffee as an example

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ABSTRACT

Corporate Social Responsibility (CSR) and a 360 degree approach to sustainable development including environmental, social and economic aspects are of growing interest. Lavazza is engaging its efforts to ensure an integrated approach aimed at providing a holistic perspective of sustainable development. Lavazza's efforts in ecodesign and LCA activities began in 2009 as part of the company's wider CSR programme. Gradually LCA and CSR have been linked in order to create an integrated approach. Whilst LCA adopts a product perspective, at the same time it contributes to the overall CSR strategy. The coordinated work between departments, within Headquarters, promotes an integrated approach to CSR, which includes the different technical corporate aspects, where LCA is one of the specific tools available fostering continuously improved environmental management. Other examples of tools are PackageExpert, which allows the performance of simplified LCA by packaging designers and the communication of results to managers, and a CSR tool that collects CSR data, partially originating from LCA activities, and enables the calculation of requested indicators. This approach is not only useful to enhance information exchange within the company, but is also a *conditio sine qua non* of an integrated CSR strategy.

Keywords: LCA, CSR, espresso coffee, sustainability, integrated approach.

1. Introduction

Sustainable development and minimization of environmental impacts within the coffee supply chain, the second most valuable commodity worldwide, are of growing interest. This trend is also visible by the increased application of current (ISO 14040:2006; ISO 14044:2006; ISO 14064-1:2012) and new environmental management standards (ISO 14067:2013; ISO 14046:2013) which focus both on product and corporate perspectives. Lavazza's innovative approach consists in ensuring that Life Cycle Assessment (LCA) is not only used as a technical tool but is part of the overall Corporate Social Responsibility (CSR) and that the two are used complementarily for environmental management.

Lavazza's corporate sustainability strategy identifies four key areas: product sustainability, production processes, human resources valorization and the interaction with stakeholders along the value chain.

Lavazza has been active on ecodesign and LCA activities since 2009; efforts to draft the first sustainability report, according to Global Reporting Initiative (GRI) standards, are ongoing (Lavazza 2014). Furthermore, Lavazza participates in a project of the Italian Ministry for the Environment, Land and Sea aimed at measuring, reducing and eventually compensating CO_2 emissions of two main coffee products: espresso coffee capsules and moka coffee (Italian Ministry for the Environment, Land and Sea 2014). At the sector level through the SAI platform, Product Category Rules (PCR) have been defined for Carbon Footprint (International EPD® System 2014).

Lavazza developed its strategies on environmental management and sustainable development, both at policy and at product level, by using LCA and CSR as two complementary approaches with different perspectives. The corporate perspective initially concentrated within the company's boundaries (gate to gate), is projected to expand those boundaries to external stakeholders such as suppliers, clients and consumers (simplified cradle to gate for all products). The product perspective looks at a particular section of the company's supply chain, analyzing the life cycle stages of a single product (detailed cradle to grave).

Although some studies have identified weaknesses of the LCA technique, its overall evaluation is nevertheless positive (Matuszak-Flejszman 2007; Lewandowska 2011). In detail, LCA has the advantage of providing an holistic perspective which is not limited to the boundaries of the organization (design, development, production processes, energy resources consumption and waste management), but also analyses the effects of environmental policies and practices of manufacturers and suppliers, extraction, transformation and distribution of raw materials, finished products distribution, use and waste management. From this point of view, a distinction between direct and indirect environmental aspects is important. Direct environmental aspects are associated with activities, products and services over that the organization has direct control. Indirect aspects concern the potential activities over which the company has no direct control but could be expected to influence.

The LCA approach measures ecological impacts, related to both direct and indirect environmental aspects. Espresso coffee is a complex food system with a multiple-actor supply chain which involves coffee plantations worldwide, green coffee traders and exporters, packaging suppliers, the coffee manufacturer, the distribution chain, consumers, clients and finally waste disposal management.

Although the involvement of all stakeholders might initially be more cumbersome, a holistic approach will transform barriers into mutual opportunities and thus strengthen market of all stakeholders on environmental management (Furfori et al., 2012).

LCA and CSR both contribute to the continuous improvement process aimed at the minimization of environmental, social and economic impact of the company's operations. LCA methodology follows a bottom-up approach, providing ecodesign feedback to R&D and pointing out possible improvement options (e.g. for green coffee cultivation and packaging) that can be implemented by the environmental product strategy. In this way, LCA has started at R&D (bottom-up) and is now contributing to core values of the company. On the other hand, CSR follows a top-down approach, starting at the management level and embedding LCA in the corporate strategy. This double strategy allows the company to have a complete perception of its environmental performances, both at corporate and product level.

The link and information exchange between LCA (product) and CSR (corporate) are crucial and specific tools, and have been developed to facilitate the interaction between the two levels. Examples of such tools are PackageExpert, which allows the performance of simplified LCA by packaging designers and the communication of results, and a CSR tool that collects CSR data, partially originating from LCA activities, and enables the calculation of the requested indicators and graphics.

The link between LCA and CSR will be illustrated with espresso coffee as an example, showing results from the tools applied. This link is not only useful to enhance information exchange between product and corporate perspective, above all it allows alignment between the CSR strategy with the product sustainability strategy.

2. Methods

2.1. LCA of espresso coffee capsule and moka coffee

LCA is a tool which allows to assess the potential environmental impact of a product, process or service along its entire life cycle (Guinée, 2002). In collaboration with the Italian Ministry for the Environment, Land and Sea LCA has been also applied to espresso coffee (Italian Ministry for the Environment, Land and Sea 2014).

The objective of Lavazza is to combine various LCAs that are interconnected in the coffee supply chain, in order to enhance ecodesign and communication and to reduce the environmental impact related to a cup of coffee.

The functional unit is defined as one cup of espresso coffee (30 cc.), prepared with an espresso machine using capsules. In case of moka coffee the functional unit is identical, but applied to moka coffee. The espresso machine is assumed to prepare 5000 cups of coffee during its lifetime. The system boundaries of espresso coffee include green coffee cultivation, processing and transport to Italy, coffee roasting, grinding and packaging, distribution, use phase (preparation of one cup of espresso with an espresso machine) and the end of life of coffee, packaging and the espresso machine. The system boundaries of moka coffee are identical, except for the production plant which is located in Torino and the preparation of the moka coffee which takes place with an electric moka machine.

In order to facilitate the elaboration of the complex life cycle of espresso coffee (containing more than 350 process units), the LCA has been divided into four sub LCAs as represented in Figure 1: LCA of coffee, LCA of manufacturing, LCA of packaging and LCA of the espresso or moka machine. Primary data are obtained from green coffee plantations and packaging suppliers using personalized questionnaires. To conduct LCA of agricultural products is a very complex task taking into account the data acquisition, modeling and compilation but it is a fundamental step for understanding the potential environmental impacts and then establishing the basis for product ecolabelling. Different agricultural practices produce different environmental performances. The amount of chemicals is directly related to cultivation practices such as tillage rotation, density of plants, etc. The goal of the LCA study on green coffee was to establish a good correlation of the agricultural practices and potential en-

vironmental impacts of coffee, increasing the internal knowledge on environmental sustainable aspects related to the raw material.
Future updates of this research will collect a large and representative number of primary data from the biggest suppliers (in this study, the primary data come from Brazil, India, Vietnam) and show the evolution of the natural resources management as land use, new agricultural practices, lower fertilizer and chemical use.

Also inventory data of Lavazza's production plants and the use phase of the espresso or moka machine are based on primary data and direct measurements. Secondary data are obtained from scientific literature, particularly for the green coffee cultivation phase (Coltro et. al., 2006) and the Ecoinvent database v2.2. Greenhouse gas emissions and other impact categories are quantified using IPCC 2007 (IPCC, 2007) and ReCiPe (Goedkoop et. al., 2008). The LCA has been conducted in line with ISO 14040/14044 (ISO, 2006), using SimaPro software (PRé, 2013).

At the product level LCA is used for ecodesign, hot spot analysis and environmental communication; whilst at the corporate level, LCA is used both strategically, to align CSR with product sustainability, and operationally, to provide a scientific basis for environmental data collection in a life cycle perspective, feeding tools such as the PackageExpert and the CSR tool.

2.2. PackageExpert

PackageExpert is a simplified ecodesign tool, which allows corporate packaging designers to develop simplified screening LCAs of different packaging solutions, enabling comparative analysis. By inserting packaging input data, such as, components' materials and weights, typology of transport, manufacturing processes and end of life options (ISPRA, 2012), PackageExpert calculates the Carbon Footprint and the Cumulative Energy Demand of the selected packaging solution.

The link between PackageExpert and the product level consists in its use by packaging designers working on ecodesign. On the other hand, the tool is regularly updated and based on scientific LCA knowledge. The link with the corporate level is the possibility to apply PackageExpert to all packaging solutions performed in a company's production plant, providing aggregated data to the CSR tool.

2.3. CSR tool

The CSR tool is a simplified tool with a corporate approach, which allows the collection of LCA data related to the entire supply chain of all products manufactured in a certain production plant. By inputting aggregated input data, such as total amount of green coffee transported to the plant, typology of transport, total amount of coffee manufactured, total distribution and waste treatment, the CSR tool calculates the Carbon Footprint and the Cumulative Energy Demand of the entire supply chain of all products manufactured in the production plant.

The link between CSR and the product level consists in including ecodesign activities in a corporate strategy: this way, LCA is embedded in a context and becomes a core tool for environmental management. Operationally, LCA provides useful information that needs to be collected for the implementation of a CSR strategy. The link between the CSR tool and the corporate level speaks for itself: the collection of environmental, economic and social data, and their aggregation in key indicators requires a dedicated tool. The CSR tool focuses on the environmental dimension, combining and aggregating LCA data with material flows and production volumes in order to obtain relevant key environmental performance indicators.

3. Results

3.1. LCA of espresso coffee capsule and moka coffee

Greenhouse gas emissions and other impact categories are quantified using IPCC (IPCC, 2007) and ReCiPe (Goedkoop et. al., 2008), as illustrated in Figure 2. The results express the relative contribution of each life cycle stage to the total impact of one cup of espresso coffee.

The results show that the most significant impacts are generated during the upstream processes (55%-82%), a small part is caused by the core processes of the coffee manufacturer (4% - 14%), while a significant remaining part is generated during the downstream processes (16%-42%). The environmental hot spots are the green coffee cultivation (32%-70%), coffee consumption (17%-28%) and packaging (3%-19%). Overall, the LCA results appear to be consistent with other studies published on coffee (e.g. Humbert et al., 2009; TCHIBO, 2008).

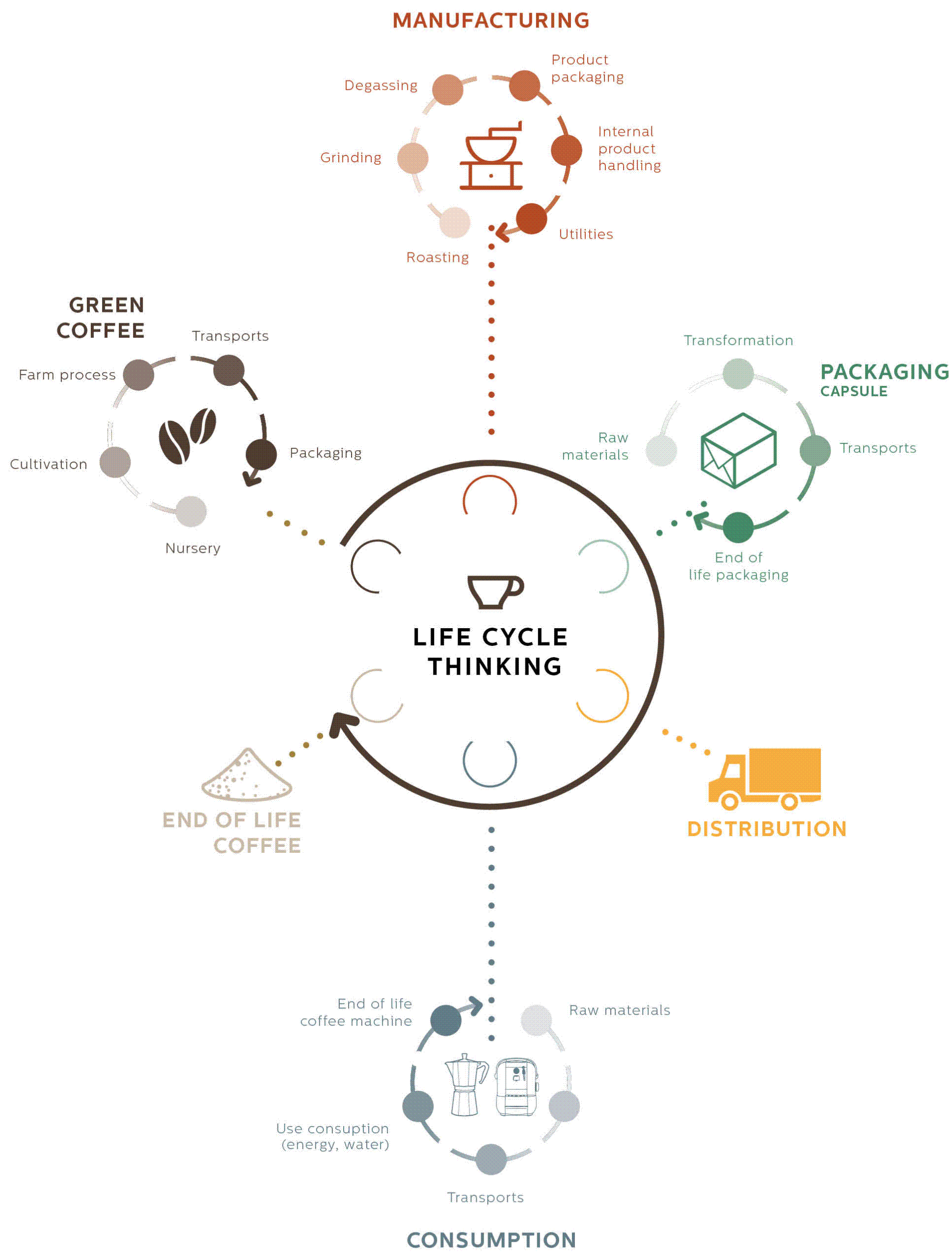

Figure 1. Combined espresso coffee LCAs .

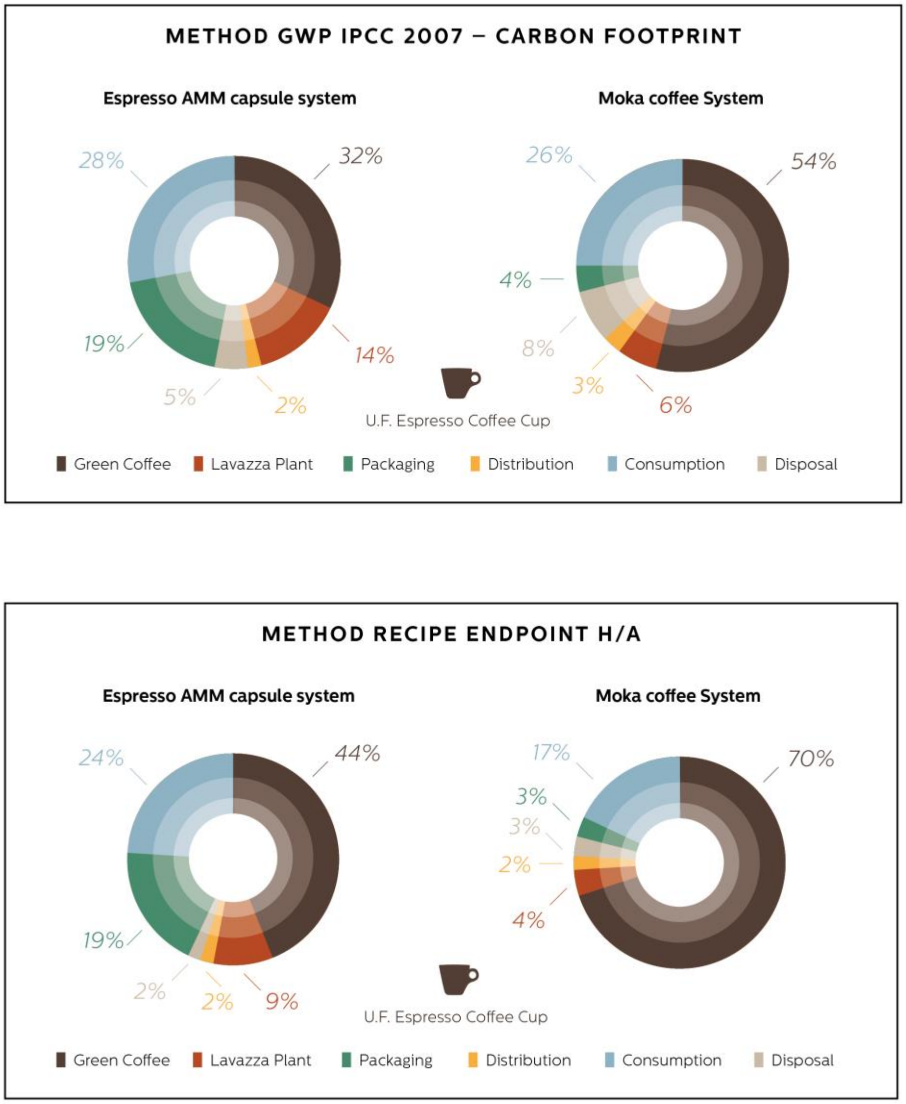

Figure 2. LCA results of one cup of espresso coffee, according to IPCC and ReCiPe.

3.2. PackageExpert

An example of the results of PackageExpert is shown in Table 1, illustrating the environmental indicators of the life cycle of a single packaging solution. Since PackageExpert is a simplified tool, it is feasible to apply the tool to all packaging solutions performed in a company's production plant. This allows to calculate not only the impact of single packaging can be evaluated (product level), but also the environmental contribution of all packaging solutions within a given production plant (corporate level).

3.3. CSR tool

The CSR tool enables aggregation of LCA data into environmental performance indicators at the corporate level. Figure 3 shows the results of the CSR tool applied to the entire supply chain of the Torino production plant, expressed in CO_2 eq. (Carbon Footprint) and MJ (CED) per life cycle stage. The CSR tool can be applied to each location and comprises besides the gate to gate analysis (processing at plant) as well as upstream (green coffee) and downstream processes (distribution, end of life). In a simplified way, based on aggregated LCA data, material flows and production volumes, the CSR tool evaluates the life cycle stages (excluding the use phase) of all main products manufactured at the production plant. In other words, it represents an aggregation of many product levels into the corporate level.

Table 1. Example of results of PackageExpert for a single packaging solution.

Packaging Life cycle	**Carbon Footprint (IPCC) [g CO_{2eq}]**	**Cumulative Energy Demand [MJ]**
Raw materials	48,5	1,291
Raw materials transport	0,8	0,014
Processing	6,5	0,131
Packaging Transport	0,5	0,009
End of life	8,0	0,008
Total	**64,3**	**1,453**

Table 2. Example of results of PackageExpert for multiple packaging solutions performed at the corporate level.

Packaging	**Amount (pieces)**	**[g CO_{2eq}]for 1 piece**	**[tCO_{2eq}] tot.**	**[MJ] for 1 piece**	**[GJ] tot.**
Packaging 1	1.000.000	115	115	2,85	2.850
Packaging 2	2.000.000	100	200	2,50	5.000
Packaging 3	1.000.000	200	200	5,00	5.000
Etc.					
Corporate			**515**		**12.850**

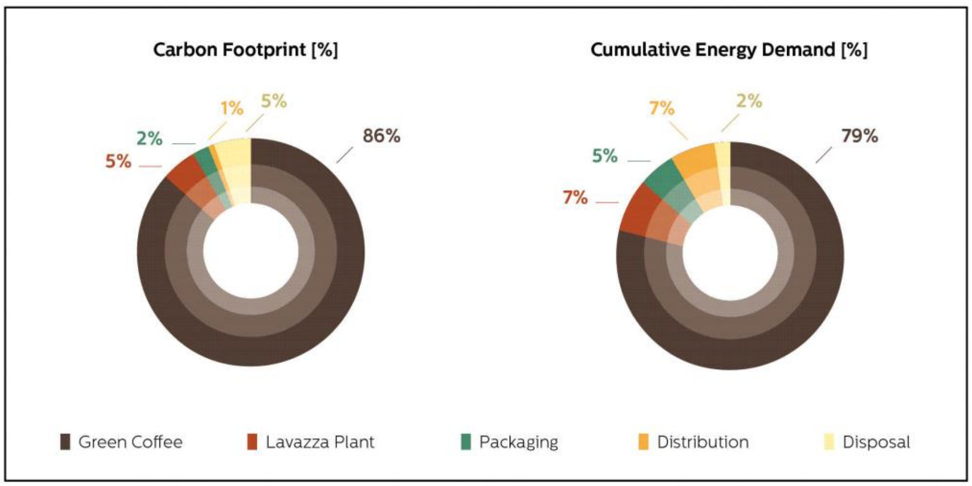

Figure 3. Results of the CSR tool for the entire supply chain of all products of the Torino production plant.

4. Discussion

LCA studies on coffee show that environmental hot spots are in particular green coffee cultivation, preparation of coffee during the use phase and packaging. In relation to the above, coffee processing (core) only generates a limited environmental impact. This emphasizes the need to view the environmental performance of coffee in a life cycle perspective (upstream and downstream).

In Lavazza, LCA was at first used in the R&D department (bottom-up) while the next step required embedding of LCA in the corporate strategy. This was achieved through CSR (top-down) thus allowing the company to have a complete perception of its environmental performances, both at product and corporate levels.

Information exchange between LCA (product) and CSR (corporate) with crucial and specific tools was developed to facilitate the interaction between the two levels. Examples of such tools are PackageExpert, which allows the performance of simplified LCA by packaging designers and the communication of results to managers, and a CSR tool that collects CSR data, partially originating from LCA activities.

Initial experiences with the different tools indicate that the tools are well interlinked, and provide added value both at the strategic and the operational levels. LCA is used strategically to align CSR with product sustainability, and, operationally, to provide a scientific basis for environmental data collection in a life cycle perspective, feeding tools like PackageExpert and the CSR tool. Besides single packaging solutions, PackageExpert can be applied to all packaging solutions of a company's production plant, providing aggregated data to the CSR tool. On the other hand, CSR takes care of embedding LCA and ecodesign activities in the corporate strategy.

The integrated LCA and CSR approach, in the context of being used for identification and assessment of environmental aspects in the corporate strategy, has advantages but also its limitations (Figure 4, SWOT). The main weaknesses, from the point of view of the considered application, includes higher time consumption and the complexity of the assessment. On the other hand, LCA enables a number of possibilities like capturing of indirect aspects, obtaining quantitative results, fostering a holistic approach through the cooperation of all stakeholders.

Figure 4. Advantages and limitations of LCA with regard to using in CSR strategy.

5. Conclusion

In conclusion, whilst an integrated LCA and CSR approach can seem more time consuming and complex to manage in terms of costs as well as unification of data, at the same time it provides a distinct advantage in terms of holistic approach, data collection, optimization and verification as well as methodology. Further, it provides an unique opportunity to achieve maximum alignment of product and corporate strategies as well as an effective stakeholder engagement.

Future work on the integrated LCA and CSR approach will focus on the improvement of the interaction between the two concepts, both at the strategic and the operational level, enhancing the information exchange between tools and systems. The obtained experience will be used to further implement this integrated approach to the entire organization of Lavazza, both at all production facilities and along the entire coffee supply chain.

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Co-products from meat processing: the allocation issue

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ABSTRACT

Allocation of the environmental burdens to the co-products generated in the meat processing sector is a methodological issue. A study was undertaken on six animal species (beef, veal, lamb, kid goat, pork, broiler) to identify the co-products generated at slaughtering and cutting stages. The status (waste or co-product) and destination of each co-product were defined. Their masses as well as their physical composition (dry matter, protein, fat and energy contents) were quantified. A sensitivity analysis was performed on seven kinds of allocation rules in order to propose the most appropriate for the meat sector: mass, dry matter content, energy, protein, fat content, a combination of the fat and protein contents, and the economic value. This led us to propose the allocation on the basis of dry matter, which corresponds to the content in fat and protein, and consequently is appropriate to reflect all the co-products functions.

Keywords: animal co-products, meat processing, allocation, life cycle assessment

1. Introduction

The environmental impact assessment of the food products through Life Cycle Assessment is a matter under development and some methodological issues are still discussed. The allocation of burdens between the various co-products of the animal after slaughtering (edible products, bone, offal not intended for human consumption, skins, etc.) is a tricky point. These products may represent until 60 % of the animal live weight and are raw materials for several economic sectors downstream (gelatine, leather, pet food, fertilization, etc.).

International guidelines and standards can help to handle this question. ISO (2006) suggests that allocation should be avoided as far as the system allows it, by (1) subdivision of the multifunctional process in sub-processes, or (2) system expansion to include the functions of the co-products. Then when allocation can’t be avoided, an allocation rule based on (4) physical causality linked to the function of the co-products must be preferred to (5) other relationships, such as economic value; (6) several of these options can be mixed. PAS 2050 (2008) follows partly the ISO standard but recommends economic allocation as the third step, if sub-division and system expansion are not feasible. The recent draft guidelines provided by the LEAP initiative, in particular on animal feed (LEAP 2014), suggest the same hierarchy as ISO and introduce a new notion: co-products are waste, residues (marginal price) or “real” co-products. Those standards recommend doing a sensitivity analysis if several allocation procedures are applicable. Most of the LCA studies identified use economic allocation (Bengtsson et al. 2012, Joseph and Nithya 2009, Notarnicola et al. 2011), at least between groups of meat co-products having the same type of usages and market. One limit of the economic allocation is that the market value of some Animal By-Products (APB) does not reflect their subsequent usage. As a consequence of animal crises in the two last decades, the ABP market prices drastically collapsed, while they still are used. Other authors also explored other possibilities. Katajajuuri et al (2008) applied an allocation on the meat-mass of the chicken co-products; Schmidt et al. (2010) for meat, and Clarke (2012) for leather prefer system expansion if possible; Bier et al. (2012) in a study on blood conclude that both allocate on mass and consider blood as waste are appropriate. The dairy industry is probably the food sector in which the issue of allocation has been studied the most. The International Dairy Federation (IDF 2010) recommends allocating raw milk and transportation on the basis of the milk solids of the final products and to allocate other inputs and emissions using a physico-chemical allocation matrix (proposed by Feitz et al. 2007). At French level, in the perspective of environmental labeling, the dairy sector has proposed to use a physical allocation based on the content of dry matter (milk solids) of the products (CNIEL and Quantis, 2012).

Then, a study was undertaken to analyze the issue of allocation for meat co-products considering six animal species: beef, veal, lamb, kid goat, pork and broiler. It aimed to better define meat co-products, explore several allocation procedures and provide recommendations for the meat sector.

2. Materials and methods

2.1. Scope of the study

The study concerns slaughtering and cutting stages in the meat processing sector, considering six animal species: beef, veal, lamb, kid goat, pork and broiler.

2.2. Status and composition of the meat co-products

The co-products generated are Edible co-products and Animal By-Product (ABP) among which ABP of C1, C2 and C3 categories are distinguished referring to the European regulation on waste (CE/1069/2009). These ABP are components presenting sanitary risks relating to conventional agents (C2) and to bovine spongiform encephalopathy (C1), proteins (PAP) C3 and fat C3, bones C3 processed in gelatine, raw fat processed into fat and “cretons”, raw skins used in tannery and organic matter used for land-spreading, composting or biogas production.

Each material was also defined as a co-product or a waste, following the recommendations of the European directive on waste (CE/98/2008). Following this regulation, ABP do not have to be considered as waste if they follow the four following criteria: (1) the later use is certain, (2) the by-product can be used without any specific treatment different from usual industrial practices, (3) the by-product is part of the production process, (4) the later use is legal.

Table 1. Co-products from the meat industry and their destination

Co-products	Destination	Direct or after process usage	Status
Meat, offal, blood, fat, rind	Food industry	Human Food	Products
SRM, sanitary seizures, residues from the first waste water treatment	Processing of ABP C1-C2 in MBM and Fat	Cement works (MBM C1), fertilizing (MBM C2), Biodiesel (Fat C1-C2)	Waste
Other meat, offal, blood, fat, rind	Processing of ABP C3 in PAP C3 and Fat C3	Pet Food, Animal Food	Products
Bones, tendons	Processing of bones C3 in gelatine	Edible gelatine; fertilizing meal	Products
Tallow, grease	Processing of raw fat C3 in fat and “cretons”	Lipochemistry, oleochemistry, fertilizers, pet food, animal food	Products
Hide, masks	Tannery	Clothes, shoes, furniture	Products
Digestive tract content, manure, residues from the first waste water treatment	Land spreading, composting, biogas	Organic matter use	Waste

SRM: Sanitary Risk Materials; MBM: Meat and Bone Meal

2.3. Allocation procedures explored and corresponding data collection

Seven kinds of allocation rules were tested, after excluding waste components. Six rules are related to physical causalities: mass, dry matter mass, energy, protein, fat content, a combination of the fat and protein contents. They were chosen to their relevance with the different functions of the meat co-products (providing feed protein, fat and energy). The last rule is the economic allocation.

To describe the slaughtering and cutting processes and the composition of the co-products obtained at each stage, a survey has been performed on 16 French plants. Additional data from literature were used to complete the data set. For each of the seven destinations identified, the mass of co-products (Table 2) as well as their physical composition (dry matter, protein, fat and energy contents) and price were quantified.

System expansion was also explored. The aim was to define a substitution scenario for each co-product and each destination. It was not always possible to define clear and objective hypothesis, as shown in Table 3, and to collect corresponding LCA data. In consequence, this way of handling co-products is not presented in this paper.

Table 2. Part of the animal live-weight (in %) to the different co-products destinations for 6 animal species (primary data from 16 processing plants in France)

Destination	Culled cow	Veal	Lamb	Kid goat	Pork	Broiler
Food industry	45%	56%	44%	44%	70%	43%
Processing of ABP C1-C2 in MBM and Fat	10%	9%	2%	2%	5%	6%
Processing of ABP C3 in PAP C3 and Fat C3	7%	6%	13%	13%	5%	51%
Processing of bones C3 in gelatine	8%	6%	3%	3%	9%	0%
Processing of raw fat C3 in fat and “cretons”	13%	6%	6%	6%	4%	0%
Tannery	6%	7%	14%	14%	0%	0%
Land spreading, composting, biogas	10%	9%	17%	17%	7%	0%
Losses	1%	1%	1%	1%	1%	0%
TOTAL	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**

Table 3. Substitution scenarios for each destination of meat co-product

Destination	Final use	Function indicator	Substitution scenario
Food industry			
Processing of ABP C1-C2 in MBM and Fat	Heat	Lower calorific value	French mix heat
Processing of ABP C3 in PAP C3 and Fat C3	Mainly pet food and animal food	fat and protein content	Other sources of fat and protein; animal or plant origin
Processing of bones C3 in gelatine	Human and animal food Fertilizer industry	Gelatine power N, P	Vegetal gelatine (algae?) Mineral fertilizer
Processing of raw fat C3 in fat and “cretons”	Human and animal food Lipochemistry	Fat content	Plant oil
Tannery	Clothes, shoes, furniture	Area (m²)	Cotton? Synthetic fabrics ?
Land spreading, composting, biogas	Land spreading, composting, biogas	N, P	Mineral fertilizer

3. Results

Figure 1 presents a selection of the allocation factors obtained with the different allocation rules, those for pork and beef (culled cows and young bulls).

For pork, from about 80% to 100% of the environmental impacts are allocated to co-products intended to human food. The impacts results of those co-products won’t vary so much through the allocation procedures tested. This is related to the fact that most of co-products are used for human food industry.

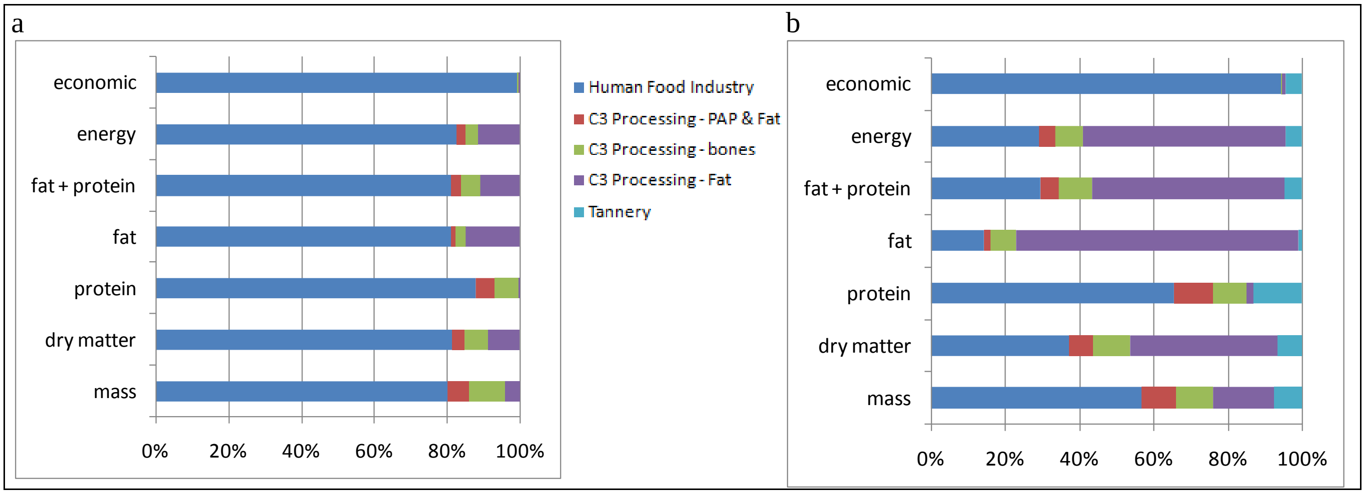

Figure 1: Distribution keys between pork (a) and beef (b) co-products (excluding wastes) according to different allocation rules

Cattle provide a wider range of different kind of co-products. For a given co-product destination, the distribution factors vary a lot from one allocation rule to another. The LCA impact results will be very sensitive to the allocation chosen. About 55% of the co-products (in mass) are valued in the food industry, while hide, bones, fat and other co-products represent a non negligible part. Economic allocation attributes almost 95% of the upstream impact to the human food industry destination and the rest to the hide. None impact is attributed to other co-products despite their downstream use. Fat and protein allocations give very contrasting allocation factors. Energy, fat + protein and dry matter procedures give almost the same allocation factors, as they all represent the interest of main of the co-products and their function in providing protein and fat.

Other species present in-between results, underlining the specificities of each of them: comparing to beef, veal factors are lower for most of the co-products, due to higher carcass yield; the relative part of tannery and C3 – PAP & Fat is higher for lamb, according to the low carcass yield and the grease component for this type of animal; for kid goats, sold as entire carcass, there are less co-products at plant gate, so a higher part is attributed to Human food industry destination; for broiler, every physical allocation procedures give about half of the impacts to human food and half to pet food (PAP & Fat).

4. Discussion

The choice of an allocation procedure should first intend to be meaningful for all co-products regarding their function. In the meat sector, most of the co-products generated are used for their physicochemical characteristics: for their content in protein (human food, pet-food) and/or in fat (gelatine, fats). That pleads for the choice of a physical allocation. A recommendation for the meat sector should be relevant for all animal species. In that way, the allocation on the dry matter content has the following advantages: this criterion combines all of the physico-chemical characteristics of interest (in particular lipids and proteins); it is relevant for the different uses and markets (food, chemistry, leather and for all animal co-products, irrespectively of their destination; it provides stable figures, few dependant of the economical context.

This approach seems appropriate for most of the European countries. Nevertheless, it is now necessary to make sure it could suit in other contexts, where animal by-products could have different status, usages and markets.

System expansion is also probably still a relevant area for future studies even if including functions of the meat co-products has limits.

5. Conclusion

The recommendation of our study is to use an allocation based on the respective dry matter content of each co-product which is relevant to consider all the functions of the co-products in their different uses and markets. It also has the advantage to be in accordance with the dairy sector. Indeed, the same approach shall be probably preferred for all animal productions from the livestock sector.

Anyway, none of the allocation procedures is perfect and these results underline the fact that a sensitivity analysis should be performed if it can have a great incidence on the impact results, such as carbon footprint. This study provides both primary data on meat processing and helpful figures for sensitivity analysis in LCA in the meat sector.

Further works, such as the actual ACYVIA program in France, would be helpful to go ahead with the allocation issue: dividing in sub processes and perhaps proposing physico-chemical allocation matrix (Feitz et al. 2007) to reflect the underlying relation between resources/impacts and the co-products. This subject should be further explored in the framework of the future Product Environment Footprint European Pilot.

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An original way of handling co-products with a biophysical approach in LCAs of livestock systems

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ABSTRACT

A new biophysical procedure to handle co-products in LCAs of livestock systems, combining subdivision of the system and allocation, has been developed. Systems were divided into animal classes, defined as dedicated to specific physiological functions (e.g. growth) and so to specific products (e.g. animals for meat). When allocation is needed, the environmental burdens of the animal class were attributed to the co-products pro rata the feed energy required to produce them (biophysical causality). This has been applied on cattle, sheep, goat, pig, poultry, rabbit and fish production systems. On the example of some dairy and pig systems, it is shown that the attribution of the burdens to the different co-products of a system changes from one impact category to another. Indeed some resource consumption or emissions are specific to an animal class and then, to a product. A sensitivity analysis, considering five other allocation procedures, allows considering their respective advantages and limits.

Keywords: life cycle assessment, allocation, livestock products, sensitivity analysis

1. Introduction

Life Cycle Assessment (LCA) results depend greatly on methodological choices. One of the most important and much debated choices concerns the allocation of the environmental flows among the co-products. At the moment, the LCA practice is not harmonized regarding the allocation of impacts to the outputs of livestock production systems. However, the question of allocation has been extensively studied, especially concerning milk production.

For dairy systems, in particular since the publication of the IDF standard (IDF 2010), most recent publications use allocation ratios based on feed-energy (Flysjö et al. 2011; Dollé and Gac 2012; Thoma et al. 2013) or feed energy and protein requirements (Cederberg and Mattsson 2000; Basset-Mens et al. 2009; O'Brien et al. 2012) needed to produce milk and animals (live weight). Depending on the authors, this procedure is called biological, physical or biophysical causality allocation. The principle is always to consider the strong causal relationship between physiological requirements, especially for growth and lactation, and the feed use for the production of live-weight and milk. From an author to another, some differences appear in the way it is applied: while IDF (2010) and Thoma et al. (2013) only consider growth and lactation needs and propose a meat/milk ratio, Dollé and Gac (2012) include five requirements, i.e. maintenance, activity, growth, lactation and gestation. Those authors highlighted that choices regarding which physiological requirements are considered could lead to significant differences of allocation ratios for a French dairy system, with 82% of impacts attributed to milk applying the IDF method, to 73% when the five types of requirements are considered. For suckler-beef production systems, it seems that only Nguyen et al. (2012) applied an allocation procedure (comparing allocation on live weight mass, protein mass and economic value through a sensitivity analysis), while other studies did not allocate impacts to the different types of animal produced. Most studies report impacts for an average live-weight, mixing cull cows, young bulls and finishing heifers, even weaners, without considering they provide different qualities of meat and that some of those animals will be slaughtered directly while others will be finished. Looking at the current literature, the situation is the same for other species, such as pork and poultry.

Nevertheless, biophysical allocation seems also relevant for other livestock production systems, especially for beef systems, because it allows having the same approach for dairy and suckler beef systems that both provide beef. When considering the allocation of impacts to the different co-products, the appropriate approach is to refer to the ISO 14044 (2006) hierarchy. First, ISO 14044 suggests that allocation should be avoided as far as the system allows it, by subdivision of the multifunctional process in sub-processes, so that inputs and outputs can be assigned to specific products. When subdivision is not possible, an allocation rule based on physical causality must be preferred to other relationships, such as economic value.

The AGRIBALYSE program has provided a public database of Life Cycle Inventories (LCIs) of French agricultural products at the farm gate (Colomb et al. 2014) according to a common methodology (Koch and Salou, 2014). The reports and summary factsheets of the AGRIBALYSE program are available on line (www.ademe.fr/agribalyse-en), while the full database is available free of charge on demand to ADEME (agribalyse@ademe.fr). Concerning animal production, AGRIBALYSE produced: (1) LCIs for animal systems that provide several co-products (e.g.: milk / cull cow / calf) and (2) LCIs of systems that provide both agricultural products (e.g.: cull sow) and living animals used as inputs in other systems (e.g.: cattle weaners, piglets for fattening units). One of the objectives of the project was to harmonize methodological choices as much as possible between all animal species. In fact, concerning allocation, the aim was to apply the same approach for cattle, sheep, goat, pig, poultry, rabbit and fish production systems.

The purpose of the paper is to present how co-products were handled in the AGRIBALYSE program, outlining amongst others the specific data collection procedure. First, we divided the processes and then we applied a biophysical allocation. Our results were compared to LCA results calculated according to other allocation methods in a sensitivity analysis. The paper focuses on the example of milk / live animals in bovine production but also provides highlights for other systems when relevant.

2. Methods

2.1. The handling of co-products in the AGRIBALYSE program

The biophysical allocation method is based on the causal relationship between feed requirements and the different products of a system: production of milk (or egg, living animal's muscles, wool) and its energy content, are driven by the energy required to support the function of lactation (respectively pregnancy, growth and wool production). Six main functions are identified: maintenance, activity, growth, pregnancy, lactation / egg production, production of wool. These functions are directly related to animal products. In current livestock production systems, most of the time, animal life stages correspond directly to different functions and so, to the corresponding products. Indeed, in all species, females become mature and productive once they have finished the main part of their growth and are then dedicated to the production of milk, calves, piglets, lambs or eggs.

This led us to consider that some life stages of an animal can be attributed to a co-product. Then, life-stages are considered as sub-processes of the whole livestock farming system. In accordance with ISO (2006), allocation can then be avoided by dividing the animal life cycle in several stages that we call "animal classes" corresponding to a characteristic physiological stage (calf to weanling, heifers, milking cows, finishing cows). Inputs and outputs were then assessed for each animal class. The output product of an animal class is a living animal that enters the following animal class, or a sold product (e.g. heifers for replacement in another herd). Figure 1 illustrates this principle for a dairy herd, making the assumption that the growth of the milk cows can be neglected.

Data were collected through a specific tool which allowed splitting information between animal classes (calf, heifers, cows…). Some data were directly available at the animal class level (feed, manure), others were available at the herd or farm scales (energy consumption). These data were attributed to animal classes using technical references (e.g. pro rata for the livestock units). When an animal class yielded a single product, allocation was not needed. When allocation was required (e.g. for the stage "milking cow": milk / calf), this was done pro rata for the estimated metabolic energy required for the various physiological functions of the animal and to produce each co-product. All requirements for the different functions were considered and not only those directly linked to a function yielding a product. Then, for the milking cows, part of the maintenance and activity functions was allocated to milk, while the other part was allocated to the calf.

The procedure developed to handle co-products in livestock systems is then a combination of division of the system and allocation.

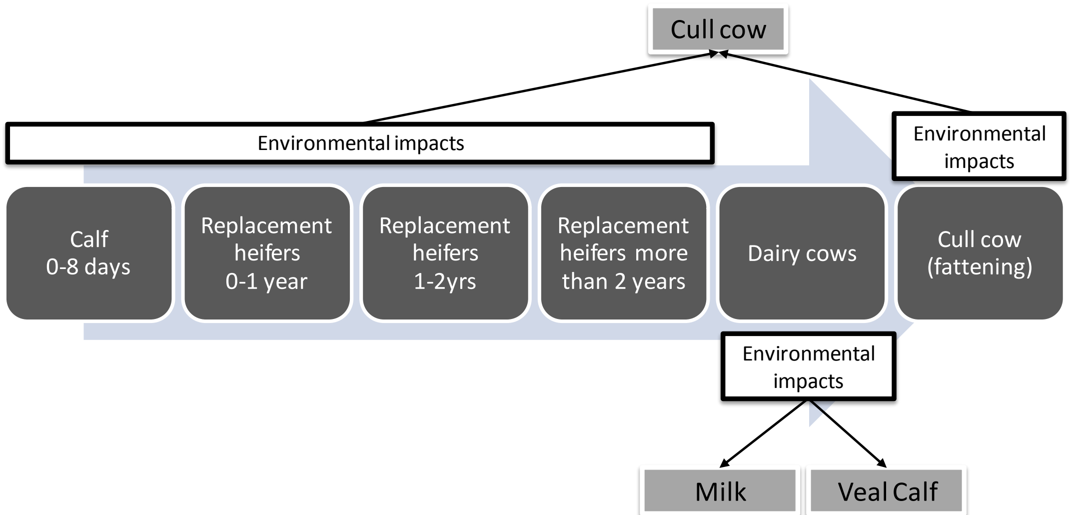

Figure 1. Allocation of inputs and impacts to co-products using a "bio-physical" representation of a dairy herd (blue: animal classes allocated to the cull cow; green: animal class allocated to milk and calves).

Table 1 provides an extract of how animal classes and their estimated requirements for the different functions were attributed to the different co-products. The energy requirements were calculated using equations from NRC (1996, 2001) for cattle in accordance with IPCC (2006) for enteric methane assessment, and national models for pork and poultry (Sauvant et al. 2004). All details are available in Koch and Salou (2014).

Table 1. Animal classes, output products and physiological functions, for allocating environmental impacts depending on the energy required for these functions.

Systems	Animal class	Output products	Physiological functions						Comments / Formulae
			Maintenance	Activity	Growth	Lactation / Egg	Gestation	Wool	
Dairy systems	Calf 0-8 days	Veal calf	X	X	X				
	Rplt Heifer 0-1yr	Cull cow	X	X	X				
	Rplt Heifer 1-2 yr	Cull cow	X	X	X				
	Rplt Heifer +2 yr	Cull cow	X	X	X				
	Milking cow	Milk	X	X		X			$Lactation + (Maintenance + Activity) * \left(1 - \frac{Gestation}{Lactation}\right)$
		Veal calf at birth	X	X			X		$Gestation + (Maintenance + Activity) * \left(\frac{Gestation}{Lactation}\right)$
	Finishing cow	Cull cow	X	X	X				
Beef systems	Calf 0-weanling	Weaner	X	X	X				
	Rplt Heifer 0-1yr	Cull cow	X	X	X				
	Rplt Heifer 1-2 yr	Cull cow	X	X	X				
	Rplt Heifer +2 yr	Cull cow	X	X	X				
	Suckler cow	Weaner	X	X		X	X		
	Finishing cow	Cull cow	X	X	X				
Pigs	Sows and piglets	Cull sows	X	X	X				
		Pig for pork				X	X		
	Post weaning	Pig for pork	X	X	X				
	Fattening	Pig for pork	X	X	X				
Layers	Chick –repro	Cull hen	X	X	X				
	Hen – repro	Cull hen	X	X			X		
	Chicken	Cull hen	X	X	X				
	Layers	Egg	X	X		X			

2.2. Sensitivity analysis

A sensitivity analysis comparing the proposed biophysical method with other methods for co-products handling was carried out on one of the five dairy systems assessed in the program (lowland, maize + grass) to analyze the consequences in terms of allocation factors and climate change impact values obtained. Six allocation methods were tested.

System expansion is the first step suggested by ISO (2006). It was performed considering that the meat from dairy cull cows and from dairy calves replace the average meat produced by the suckler and finishing beef systems analyzed in the AGRIBALYSE program (a mix of suckler cows, heifer and young bulls: 16.23 kg CO_2 eq./kg LW).

Economic allocation was done using data from the case study which provided the price of milk and animals sold for the year 2008 (Réseaux d'Elevage, 2009).

For the protein allocation, protein production by milk and animals were derived from the production of the farm and the average amount of protein in milk (33.2 g/liter) and the amount of protein per kg of live weight (150 g/kg) (CORPEN 1999).

The IDF allocation factor was calculated following the equation provided (IDF 2010):

$$R = 1 - 5.7717 * (\text{meat} / \text{milk}) \quad \text{Eq. 1}$$

The biophysical allocation method proposed by Dollé and Gac (2012) for dairy herds helps to consider the five functions in a simple way: all the functions of milking cows are allocated to milk, except pregnancy dedicated to the sold veal calves and all the functions of replacement heifers are allocated to the cull cow (including pregnancy, considering it contributes to the replacement calves).

Another allocation rule is added, derived from the AGRIBALYSE procedure, where energy requirements are attributed to the co-products in the same way (Table 1) to establish allocation factors (cull cows: all requirements of the replacement heifers; veal calves: requirements for pregnancy of the cows and a part of their maintenance and activity, plus the requirements of the dairy calf until they're sold; milk: requirements for lactation of the cows and a part of their maintenance and activity).

3. Results

3.1. Consequences of the AGRIBALYSE procedure on allocation factors

A selection of results in Table 2 presents how the impacts are distributed to each co-products for the five dairy systems studied in the AGRIBALYSE program. The resulting factors vary from one system to another, depending on its technical performances (milk production per cow, replacement rate, etc.). They also change from one impact category to another, because some impacts are specifically linked to one stage of production. For instance, considering milk production, the highest factor is always the one for non renewable energy, because of the energy consumption of the milking parlor.

Table 2. Distribution of the impacts with the AGRIBALYSE biophysical procedure for two impact categories and five dairy systems.

Impact category		**IPCC GWP 100a**			**Non renewable Energy fossil + nuclear**		
Co-products		Calf	Cull cow	Milk	Calf	Cull cow	Milk
Functional unit		1 kg LW	1 kg LW	1 kg FPCM	1 kg LW	1 kg LW	1 kg FPCM
Dairy systems	Lowland - Maize	1.5%	17.8%	80.7%	1.6%	12.9%	85.5%
	Lowland - Maize + Grass	1.5%	22.9%	75.6%	1.7%	15.7%	82.6%
	Lowland - Grass	1.9%	29.2%	68.9%	3.1%	17.1%	79.8%
	Lowland - Grass - Organic	2.0%	23.6%	74.4%	2.3%	15.0%	82.7%
	Highland	2.6%	19.7%	77.7%	3.0%	8.8%	88.2%

LW: Live Weight; FPCM: Fat and Protein Corrected Milk.

Amongst the co-products of a whole pig production system (breeding and fattening), the impacts are mainly supported by the fattened pigs, while cull sows carry less burdens. It is due to the fact that nearly 20 pigs are produced per sow per year. The fattening pigs are the product that the farms intend to produce and the sows are a tool of production even if their meat is valorized. Two contrasting types of production (conventional and organic) (Table 3) show the high allocation factors for fattening pigs for the impact climate change (86% and 92%). The allocation factor for fattening pigs is higher for organic production because the housing mode of the pigs is on litter. More GHG are emitted than on the slatted floor of the conventional production. Despites this high allocation factor for fattening pigs, the impact values per kg live animal are higher for culled sows than for fattening pigs (11.54 kg CO_2 eq. / kg of culled sow *vs* 2.4 kg CO_2 eq. / kg of fattening pig). This is due to the live weight production of each animal class. This allocation factor doesn't reflect the economic value of the products, respectively 93.5% of the income coming from pig sales and 6.6% from sows.

Table 3. Biophysical allocation factors obtained for two impact categories and two pig systems.

Impact category		**IPCC GWP 100a**		**Non renewable Energy fossil + nuclear**	
Co-products		Pig for pork	Culled sow	Pig for pork	Culled sow
Functional unit		1 kg LW	1 kg LW	1 kg LW	1 kg LW
Pig production	Organic	91.8%	8.2%	84.4%	15.6%
	Conventional	85.6%	14.4%	85.3%	14.7%

3.2. Sensitivity analysis results

The sensitivity analysis performed shows that the carbon footprint (CF) of the milk in the maize + grass system varied from 0.70 to 1.14 kg CO_2 eq./ kg FPCM (Table 4) depending on the co-product handling. These results are in the range of values from the literature. The CF varies from -27% to +17% comparing with the value provided by AGRIBALYSE. It has much more consequences on the impact of the carbon footprint of live weight animals, dedicated to produce meat directly (cull cows) or to be finished (calf): from -54% to +110%.

Table 4. Effect of the co-product handling procedure on allocation factors (in %) and carbon footprints (CF) of milk, cull cow and veal calf for a French lowland dairy system based on silage maize and grass dairy system (in kg CO_2 eq. per kg of FPCM and per kg of LW)

	Veal calf		Cull cow		Milk	
	factor	CF	factor	CF	factor	CF
Economic allocation	1.2%	6.10	10.4%	4.01	88.4%	1.14
Protein allocation	1.0%	5.01	13.0%	5.01	86.0%	1.11
Biophysical allocation (IDF)	1.5%	7.45	19.4%	7.45	79.1%	1.02
Biophysical allocation (Dollé & Gac, 2012)	2.9%	14.71	19.3%	7.40	77.8%	1.00
Biophysical allocation – derived from AGRIBALYSE	5.3%	27.02	19.3%	7.40	75.4%	0.97
Division + allocation - ABRIBALYSE® -	1.5%	7.73	22.9%	8.80	75.6%	0.98
System expansion	-	16.23	-	16.23	-	0.70

4. Discussion

Advantages and limits of each allocation procedure tested are discussed here. In our example, economic allocation provides the highest carbon footprint of the milk. It allows attributing impacts to each co-product, related to its market value. It shows the real interest for each of the co-products. However, the economic value of a dairy calf at the dairy farm gate does not represent the final use of this co-product which continues its life cycle in a finishing unit. To exclude the effects of short-term price variability, economic allocation should be done using average prices based on data for several years, however, this has not been possible here. Another point concerning economic allocation is that it could be not relevant when comparing productions from different countries, with different races. The example of sheep production in France and New Zealand is relevant: in France, wool provides little economic profit, while in NZ, because of the more productive races and an existing market for carpet making, the wool has a good value (Gac et al. 2012).

Protein allocation is oriented by the protein content of each co-product leaving the farm, in relation to their destination as human food. It doesn't help distinguishing cull cows and calves as they have both the same functional unit (kg of live weight). The results of carbon footprints are close to those obtained with the economic approach.

The IDF (2010) method provides also the same results for cull cows and calves. That is explained by the fact that this physical allocation method is based on a milk/meat ratio which treats all animals dedicated to meat production in the same way. This is probably linked to the fact that this allocation method was developed specifically for the dairy sector and didn't aim at specifying each co-product.

The biophysical allocation methods of Dollé and Gac (2012) and the one derived from AGRIBALYSE assign less energy requirements to milk. This results in a lower carbon footprint for milk and a higher carbon footprint for live animals, in particular for the veal calf. When those allocation procedures are applied, the carbon footprint of dairy animals at the farm gate is much closer to that of animals from suckler beef systems. This point is relevant because these two types of beef are sold in the same market. Beef from dairy cows is important, as it represents 23.5% of the French beef production (Institut de l'Elevage, 2013). This method helps also to properly consider French mixed races (Normande, Montbelliarde) which produce less milk and more meat than specialized milk breeds such as Holstein Frisian.

System expansion yielded the lowest carbon footprint of milk. It considered that beef from dairy herds is comparable to other types of beef. This is based on the fact that beef from culled dairy cows is sold in the same markets and for the same uses (mainly human food), despite their difference in terms of quality and price. However, in the way we applied system expansion, it was not possible to distinguish the kilograms of live weight of cull cows and of calves: the former is dedicated to being slaughtered, while the latter will be finished in specific farming units. System expansion should have included finishing of dairy calves and the comparison with suckler calves to be consistent regarding all three co-products (milk, dairy calves, cull cows). Anyway, this way of handling co-products provides impact values for the main product (milk), but does not provide impact values for the other co-products. System expansion could be applied on a dairy system to determine the impacts of calves and cull cows by substituting cow milk by milk of other species (goat, sheep). However, this is quite an unrealistic hypothesis, since goat and sheep milk really serve different markets and needs than cow milk (mainly cheese processes). This example illustrates a weak point of the system expansion approach, as finding an equivalent product is often difficult. Furthermore the system expansion approach is quite subjective because the result is closely dependant on the scenario of substitution that was chosen.

Finally, the procedure applied in the AGRIBALYSE program provides intermediate results, both for milk and for living animals. The corresponding allocation factors are very close from those of Dollé and Gac (2012) for carbon footprint, due to similar methodology: the main difference is the attribution of the maintenance and activity energy needed by cows which are there distributed between milk and calf. However, the allocation factors can deviate when considering other impacts (Table 2). Indeed, the specificity of this approach is that it provides allocation factors specifically for each impact category. This makes sense because, even if feed intake is the main driver of animal excretion and most of the emissions and impacts (climate change, acidification, water use), but this is not the case for all impact categories, specifically for energy demand and land occupation, and eutrophication to a less extent.

5. Conclusion

This paper proposes a new way to handle animal co-products at farm gate, based on a bio-physical approach, coupling subdivision of the system and allocation for the first time. This method is relevant for all livestock systems; it is applicable either for milk, meat, egg or wool. It is also relevant for all impact categories, whenever if they are led or not by animals consumptions and excretions. System subdivision allows determining environmental impacts at for each animal development stage, which can be useful to consider impacts of every type of living animal at the farm gate, even if they don't have the same destination: slaughtering, finishing or replacement. In this way, inputs and emissions are directly attributed to the corresponding stage and product. From the farmer's perspective, this approach would help to reconsider the multifunctionality of the production even on specialized systems. Mitigation options or practices improvements would then either be chosen to be adopted on the whole farm or to firstly lower the environmental footprint of the main product.

However, this method requires an appropriate data collection tool and detailed primary data per animal class, which can be quite difficult to obtain. If this is not possible, the alternative is the application of biophysical allocation factors, such as the one proposed derived from the AGRIBALYSE procedure, uniformly on the different impact categories. This first application at a large scale, should know probably be improved by using national models for feed requirements and the way in which the physiological functions are attributed to the different co-products is open for discussion and for further improvement.

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Effect of on-farm biogas production on impacts of pig production in Brittany, France

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ABSTRACT

In the context of climate change and non-renewable energy depletion, the transition toward increasing the contribution of renewable energy requires insight into environmental consequences of such energy production, such as anaerobic digestion (AD). In a pig farm producing crops used as ingredients for pig feed, biomass (fertilized with pig manure), crop residues and intercrops are highly valuable substrates for AD. Thus, the aim of this work was to assess the influence of on-farm digestion of pig slurry to produce bio-energy on environmental impacts of pig production from a life cycle perspective. This system allows maximum autonomy for the farmer, regardless of the availability of digester nutrients. It had lower negative impacts for most impact categories than a more energy-dependent system, even for hotspot impacts of climate change and cumulative energy demand. More accurate data about carbon mineralization of digestates are necessary to make conclusions about potential impacts on soil organic matter dynamics.

Keywords: anaerobic digestion, pig production, soil organic matter change

1. Introduction

In the context of climate change and non-renewable energy depletion, the demand for renewable energy is rising, and the European Union aims to obtain 20% of its energy from renewable sources by 2020 (EU 2009). This transition requires insight into environmental consequences, such as climate change (CC) and cumulative energy demand (CED), of renewable energy production. Change in soil organic matter (SOM) is also a hotspot indicator for soil quality (Garrigues et al. 2012) in the case of bio-energy produced from biomass by anaerobic digestion (AD) in the form of heat and electricity. The product that remains after AD is digestate, which can be recycled as organic fertilizer for crop cultivation. On a pig farm in which the main crops used as ingredients for pig feed are produced on the farm, biomass (fertilized with pig manure), crop residues and intercrops are highly valuable substrates for AD. Thus, the aim of this work was to assess the influence of on-farm co-digestion of pig slurry to produce bio-energy on environmental impacts of pig production from a life cycle assessment (LCA) perspective.

2. Methods

2.1. System definition

The agricultural system is assessed via its function of food production. System boundaries of the pig breeder/fattener system are from cradle to the farm gate. The system includes infrastructure, inputs, related resources and emissions for pig production and the crops used as ingredients in the pig feed. For crops used as pig-feed ingredients, the main crop rotation in its region of origin was assumed. For crops produced on the pig farm, crop rotations were considered in greater detail. Temporal system boundaries of crop rotations end with the harvest of the crop analyzed and begin after the harvest of the preceding crop. The functional unit is 1 kg of pig liveweight produced. Impact categories assessed were those of the CML-IA, plus CED and two soil-quality indicators: compaction and SOM change (Garrigues et al. 2012).

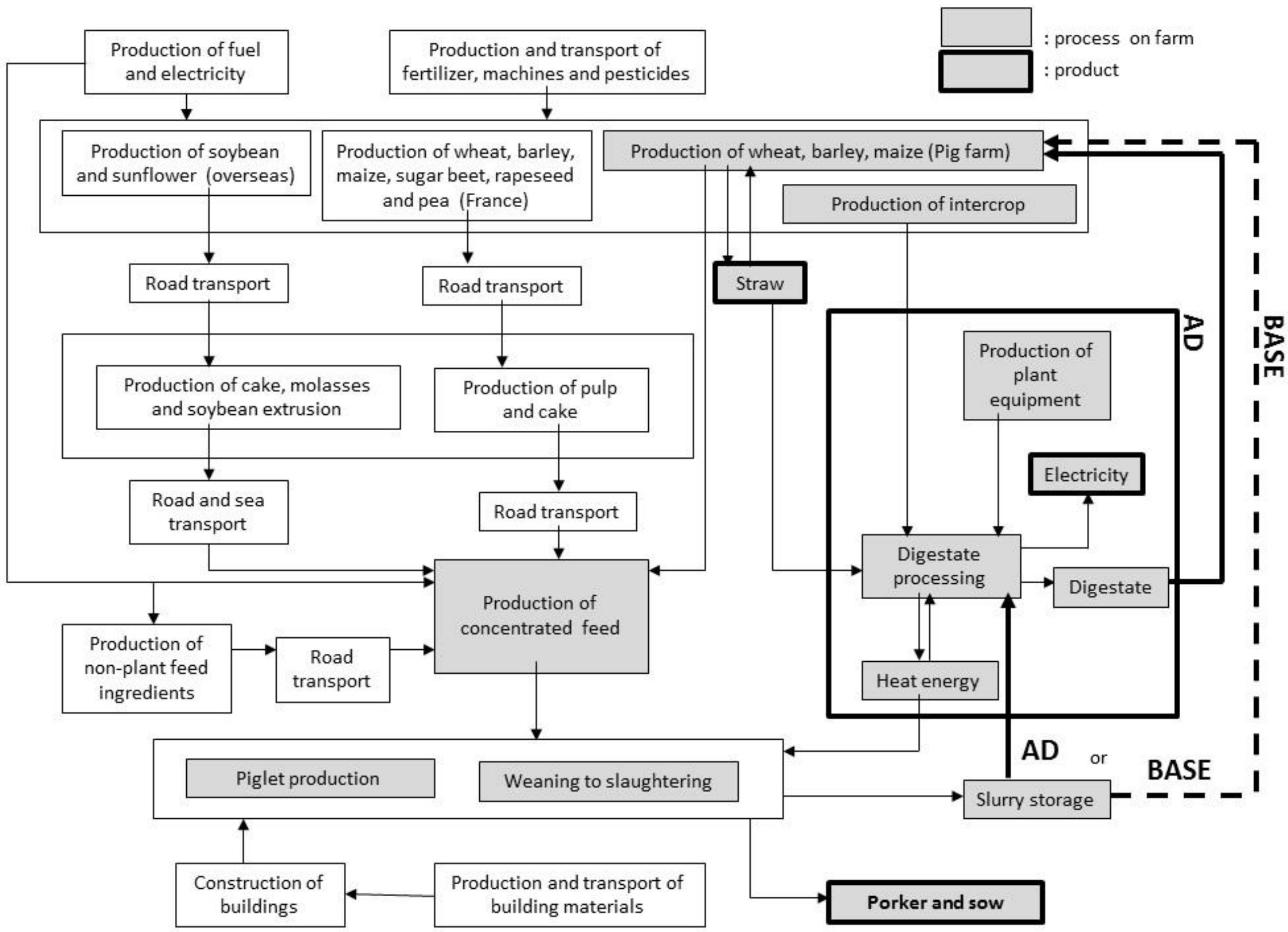

Figure 1. System boundaries for baseline (BASE) and anaerobic digestion (AD) scenarios

2.2. Definition of scenarios

We compared two scenarios (Fig. 1): (1) standard manure storage and spreading on a representative pig farm in Brittany, France (BASE scenario), and (2) the same pig farm with a 50 kW AD plant with digestate spreading on fields instead of slurry (AD scenario). The representative farm produces 4800 pigs per year with 225 permanent sows. Maize, wheat and barley are produced on 115 ha with phacelia as an intercrop. Four rotations are represented: wheat-maize and wheat-barley-maize on 48 ha each, wheat-maize-maize on 15 ha and maize monoculture on 4 ha. In the AD scenario, oats and triticale are also grown as intercrops with phacelia to feed the AD plant (Table 2). Fertilizer applications (mineral and organic) were calculated to maintain yield and to prevent pollution risk. Characteristics of the pig farm (e.g., volume of pig production, area, crops) were based on examples of existing pig farms in Brittany that remained economically viable after adding AD to the farm (Brittany Chamber of Agriculture, pers. comm.).

2.3. Life cycle inventory assumptions and model used for SOM change

2.3.1. Composition of pig feed in the two scenarios

Pig production in the breeder/fattener system uses seven different feeds, with compositions defined by the French Pork and Pig Institute (IFIP) according to the pig-production stage (Table 1). The farmer fabricates all feeds on the farm. All crop production is used to feed the sows, piglets and pigs and represents 88% of the maize, 82% of the barley and 70% of the wheat needed for the feeds. The rest of the ingredients are bought on national and international markets (Table 1).

Table 1. Ingredient composition (percentage by mass) and sources of representative pig feed produced in Brittany.

Ingredient	Maize	Wheat	Wheat middlings	Barley	Rape-seed oil	Rapeseed cake	Soya cake	Sunflower cake	Sugarbeet pulp
Origin	France (Brittany)	France (Brittany)		France (Brittany)	France (Brittany)		Brazil (Santa Catarina)	Argentina (Balcarce)	France (Picardy)
Source crop	Maize	Winter wheat		Barley	Rapeseed		Soya	Sunflower	Sugarbeet
Economic allocation[a] (%)	100.0	100.0	14.0	100.0	75.6	24.4	66.0	37.6	15.6
Pregnant sow feed (%)	6.8	40.0	3.2	30.0	0	6.7	0	5.0	5.0
Lactating sow feed (%)	12.7	30.0	0	32.0	0	5.7	13.0	2.0	1.0
Piglet prestarter feed (%)	0	61.9	0	0	4.6	0	27.6	0	0
Piglet starter feed (%)	0	42.0	0	32.0	0	0	20.0	0	0
Post-weaning feed (%)	0	58.0	0	20.0	0	0	19.0	0	0
Growing pig feed (%)	20.0	43.0	0	15.0	0	0	19.0	0	0
Finishing pig feed (%)	50.1	28.0	0	0	0	0	17.0	0	0

[a] Economic allocation based on Olympic mean price from 2006-2010 (ISTA 2009 & 2011).

2.3.2. AD scenario

In seeking maximum autonomy for the farmer, dimensions of the AD plant (energy produced and substrate quantities needed) aim for economic viability of the pig farm: no need to import substrates for AD other than those produced on the farm and spreading of all digestate produced on farm crops as fertilizer.

The AD is operated at a mesophilic temperature (around 35°C) with a hydraulic retention time of 66 days. CH_4 production from the AD is 110,832 m^3/year, with an electricity-production capacity of 50 kW and energy efficiency of 36%. Operating the AD requires 3% of the electricity produced and consumes 36% of the heat produced. The heat produced covers all heating needs of pig buildings and provides a surplus in summer. This scenario reflects simple biogas installations, which produce less electricity but also cost less to install.

Table 2. Substrates of the anaerobic digestion (AD) plant and their organic matter (OM) before and after AD

Substrates	% of substrate produced on-farm used for digestion	t/year	OM before digestion (t)	OM after digestion and storage (t)
Pig slurry	100	207	147	100
Wheat chaff	100	71	65	31
Barley chaff	100	17	15	7
Grass silage	100	12	11	4
Oats (intercrop)	100	56	53	9
Triticale (intercrop)	100	60	56	13
Barley straw	50	28	26	15
Maize stalks	33	64	59	34

To obtain a liquid rather than solid digestate, not all of the available straw and maize stalks are placed into the digester. The digestate is liquid enough (5.8% DM) to be spread as fertilizer, in the same way that slurry is spread in the BASE scenario. In the AD scenario, slurry and digestate are stored separately in covered concrete tanks, and the remaining 50% of the wheat straw is sold. CH_4 and N_2O emissions of the AD plant (storage and digestion) were calculated with the DIGES tool (Gac et al. 2006). Since emission factors for digestate applied to fields were not available, we assumed them to be the same as those for pig slurry. In the BASE scenario, slurry is stored in and open concrete tank, and 100% of the wheat straw is sold. During storage, nutrient leaching from the slurry or digestate is assumed to be negligible in both scenarios.

2.3.3. Soil carbon change modeling

RothC (version 26.3) simulates dynamics of organic carbon (C) in soil (Coleman *et al.* 1997; Coleman and Jenkinson 2008). The effects of soil type, temperature, moisture content and plant cover are considered in the turnover process. Soil organic C is split into four active compartments and a small amount of inert organic matter (IOM). The four active compartments are decomposable plant material (DPM), resistant plant material (RPM), microbial biomass (BIO) and humified organic matter (HUM). When exogenous organic matter is added to the soil, it is split between the DPM, RPM and possibly HUM pools according to partition coefficients, such as DPM/RPM. For crop residues, we used a DPM/RPM ratio of 1.44, i.e. DPM of 59% and RPM of 41%. For slurry and digestate, DPM and RPM was calculated from their Van Soest biochemical fractions via an indicator of remaining organic C (I_{ROC}) (Lashermes et al. 2009) based on equations developed by Peltre et al. (2012):

$$DPM = -1.254 \times I_{ROC} + 115.922 \qquad \text{Eq. 1}$$

$$RPM = 0.979 \times I_{ROC} - 8.928 \qquad \text{Eq. 2}$$

We used DPM of 63.9% and RPM of 31.7% for slurry and DPM of 29.5% and RPM of 58.5% for digestate.

We simulated 20 years of the same management practice and divided the total change in soil organic C by 20 to provide the mean rate over one year. When analyzing crop rotations, temporal system boundaries differed from those of individual crops, for which the C brought to the soil by crop residues was considered from just after the harvest of the crop of interest (when residues are left in the soil) to the harvest of the following crop. We thus followed back-effects of the C supplied. The change in soil C could be positive or negative, indicating soil C storage or loss, respectively.

3. Results

Table 3. Potential impacts of the (BASE)line and anerobic digestion (AD) scenarios per kg of pig liveweight according to the CML-IA method.

Impact category	Unit	BASE	AD	AD vs. BASE (%)
Abiotic depletion	kg Sb eq	3.96E-03	3.66E-03	-7.6
Acidification	kg SO_2 eq	5.45E-02	5.57E-02	+2.1
Eutrophication	kg PO_4^{---} eq	1.53E-02	1.57E-02	+2.5
Climate change (GWP100)	kg CO_2 eq	1.90E+00	1.86E+00	-1.9
Ozone layer depletion	kg CFC-11 eq	9.07E-08	8.66E-08	-4.5
Human toxicity	kg 1.4-DB eq	9.36E-01	9.21E-01	-1.6
Fresh water aquatic ecotoxicity	kg 1.4-DB eq	9.33E-01	9.22E-01	-1.2
Marine aquatic ecotoxicity	kg 1.4-DB eq	3.65E+02	3.58E+02	-2.0
Terrestrial ecotoxicity	kg 1.4-DB eq	1.52E-01	1.50E-01	-1.1
Photochemical oxidation	kg C_2H_4	8.81E-04	8.79E-04	-0.3
Land occupation	m^2y	3.93E+00	3.94E+00	+0.2
Total cumulative energy demand	MJ	1.18E+01	1.09E+01	-8.1
Soil organic matter change	kg C	1.35E+01	1.12E+01	-17.5
Compaction	m^3	1.21E+01	1.90E+01	+36.4

LCA of the two scenarios showed lower environmental impacts per kg of pig liveweight of the farm with AD (Table 3). Installation of an AD plant reduced CC by 1.9% (BASE: 1.90 kg CO_2-eq/kg; AD: 1.86 kg CO_2-eq/kg) and CED by 8.1% (BASE: 11.84 MJ/kg; AD: 10.88 MJ/kg). SOM was sequestered in both scenarios (positive impacts) despite less soil C storage when straw was exported compared to a scenario (results not shown here) in which straw was left on the soil. The BASE scenario sequestered a mean of 13.5 kg C/kg and the AD scenario 11.2 kg C/kg. The AD scenario had higher acidification and eutrophication impacts than the BASE scenario (by 2.1 and 2.5% respectively). The greatest increase in impact due to the installation of an AD plant was for soil compaction, which increased by 36.4%. The AD scenario had lower impacts than the BASE scenario for the other categories.

4. Discussion

Introducing biogas technology in a pig farm in Brittany reduced greenhouse gas emissions, mainly by replacing natural gas for heating piglet nurseries. In France, where most electricity is produced by nuclear energy, CC impacts from electricity production were similar for both scenarios. Even though the AD plant studied was small, CED decreased due to direct production and use of heat and electricity.

The increase in SOM was lower in the AD scenario than the BASE scenario. AD's digestate has a lower C content than the substrates used to create it because of the loss of C via CH_4, but it has greater stability, which decreases long-term loss of SOM. In a Danish study (Thomsen et al. 2013), a three-pool model used to simulate

C mineralization predicted similar long-term C sequestration in soil for initial turnover of plant biomass in the soil, ruminant digestive tracts, an AD plant or a combination of the latter two. In that study, C pools in the model were calibrated with laboratory incubations. In our study, the RothC model was initialized with pedotransfer functions. Although both scenarios sequester C, the lower C sequestration due to spreading digestate instead of slurry can be compensated by changing straw management: C sequestration was 19.5 kg C/kg (74% higher) if all straw was returned to the soil in the AD scenario.

Adopting AD technology on a pig farm appears to increase compaction due to the greater number of field operations required for the new intercrops grown to feed the AD plant. Since the compaction indicator only considers neutral (plowing) or negative impacts of field passes, the more field operations, the higher the predicted compaction (Garrigues et al. 2013). Furthermore, the new operations occurred when soil water content was high (spring and autumn), thereby increasing compaction risk. The compaction indicator represents the potential impact well if these practices are performed every year. Indirect implications of soil compaction, such as lower future yields, should not be neglected, but crop rotations should decrease this risk. Despite this, AD of slurry, straw and intercrops is well accepted in France. Replacing the intercrops with crops to feed the digester would probably decrease compaction and increase efficiency of the AD, but this practice is not currently accepted in France. A sensitivity analysis is planned to assess the relative influence of farm and AD plant characteristics.

5. Conclusion

Installation of a small AD plant on a pig farm producing most of the ingredients in its animal feed provides maximum autonomy for the farmer, who does not have to depend on the availability of plant substrates for AD. The AD system tended to have lower environmental impacts than a more energy-dependent system, even for the hotspot impacts of climate change and cumulative energy demand. Careful attention must be paid to SOM management through C amendment with digestate and straw. Digestates are quite new types of exogenous organic matter with a wide diversity of C mineralization characteristics depending on the plant substrate. More accurate data about carbon mineralization of digestates are necessary to make conclusions about potential impacts on SOM dynamics.

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Assessing the socio-economic and environmental impact of GMO corn varieties and consequential changes in farm management practices

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ABSTRACT

Both herbicide-tolerant and insect-resistant GMO varieties as well as combinations thereof are now widely used in corn production in the US. The introduction of these traits has led to changes in farm management practices most notably adapted pesticide usage and adoption of no-till cultivation practice. This study assesses the social, economic and environmental impact of GMO corn cultivation in Nebraska using an LCA-based approach. The Nebraska Crop Budgets published by the University of Nebraska, Lincoln, are used as main input data. Whereas the social impact remains similar in the sum of all aspects considered, the economic and environmental impact assessment showed clear advantages of the use of GMO traits in all economic and most ecological aspects considered. Overall, the study shows that production and use of fertilizer as well as the yield achieved are the main drivers for sustainable corn production. The environmental impact of pesticide production and its adapted use through the introduction of GMO varieties is comparably limited although the economic benefits of reduced pesticide use associated with GMO corn varieties are significant.

Keywords: GMO, corn, farm management, no-till, pesticide use

1. Introduction

Corn is the dominant crop in the US grown on 95,6 mio acres in 2013 (USDA National Agricultural Statistics Service; http://www.nass.usda.gov). Corn is used mainly as a source of starch and protein rich feed, for the production of bio-ethanol and processed as high-fructose corn syrup in food production. Corn productivity has increased substantially from 72 to 153 bushels per acre in the time between 1970 and 2010 (USDA National Agricultural Statistics Service; http://www.nass.usda.gov). Since their introduction in 1996, GMO traits have been rapidly adopted by farmers in the US with roughly 90% of all corn being GMO corn in 2013.

GMO traits that are being used by farmers on a regular basis now include herbicide-tolerance for efficient weed management and insect-resistance to control major Lepidopteran pests such as the corn borer (e.g. Ostrinia nubilalis) or corn rootworm (e.g. Diabrotica sp.). In particular, herbicide-tolerant crops expressing a mutant version of the 5-enolphyruvylshikimate-3-phosphate synthase gene conferring tolerance to the application of the broadband herbicide glyphosate also known under its brand name RoundUp (RR) is widely used. Alternative herbicide-tolerance traits to be used in combination with glufosinate, dicamba or 2,4-D are available to farmers. For insect-resistance, corn expressing different versions of *Bacillus thuringiensis* toxins (Bt) in different parts of the plant or constitutively are generally used by farmers to control pests. Furthermore, a variety of stacks of different Bt traits as well as in combination with herbicide-tolerance traits such as Agrisure or SmartStax in which 8 different transgenes in one corn variety is combined are the latest in development of GMO corn varieties.

For farmers the economic benefits of GMOs in terms of better weed and pest management and resulting higher yields and profits are evident (Brookes and Barfoot 2009). An important factor for the rapid and vast adoption of GMOs in corn production also seems to be a better management of yield risks associated with for example pest infestation (Shi et al., 2013). In terms of environmental benefits, herbicide-tolerant GMO corn in the US alone is associated with a 10,9% reduction in use of pesticides or 180,2 million kg of active ingredient and insect-resistant GMO corn in the US with a 41,9% reduction or 40,7 million kg of active ingredient cumulatively in the period between 1996 and 2011 (Brookes and Barfoot 2013). The same analysis furthermore calculates that herbicide-tolerant crops in the US have saved 77 million liters of fuel due to the adoption of no-till practices and consequential changes in field operations in the year 2011 alone.

The benefits of GMO corn reported above are impressive though they represent singular benefits at farm level. For a comprehensive analysis if GMO varieties contribute to sustainable development in corn production, the social, economic and ecological impacts need to be assessed and impacts both at farm as well as within the value chain need to be considered

In this study we have applied an LCA-based methodology to assess the social, economic and environmental impacts associated with the cultivation of 4 different corn varieties in Nebraska. Whereas one corn varieties was a conventional variety produced by "classical" breeding methods, the three other varieties had either only a GMO herbicide-tolerance trait, a combination of herbicide-tolerance and insect-resistance or incorporated the current maximum of traits as in SmartStax.

Lifecycle assessment (LCA) has been used before to assess environmental and economic impact of corn production, but mostly in relation to its use as feedstock for bio-ethanol production from corn grain or stover (see for example Pieragostini et al. 2014; Whitman et al. 2011). LCA has not been applied so far to specifically assess the impact of GMO traits in corn production.

Incorporation of herbicide-tolerance and insect-resistance GMO traits into corn varieties directly translate into changes in farm management practices such as new crop protection regimes. These GMO traits also greatly favour adoption of new soil management practices such as no-till cultivation. These changes in farm management are specifically accounted for. In scenario analysis, the robustness of the model and options for further improving the sustainability of corn production are evaluated.

2. Methods

2.1. Goal and scope

The aim of this study is to assess the impact of the introduction of GMO traits on corn production in the state of Nebraska, USA. Since 1996, GMO traits conferring tolerance to the application of broad-band herbicides, mainly glyphosate or RoundUp (RR), or traits to control major Lepidopteran pests such as the corn borer (e.g. Ostrinia nubilalis) or corn rootworm (e.g. Diabrotica sp.) by expressing insecticidal proteins from Bacillus thuringiensis (Bt) in corn. In the study the production of non-GM corn is compared to GM corn with only herbicide-tolerance based on glyphosate (RR), a commonly used combination of herbicide-tolerance and insect-resistance (RR&Bt) as well as a SmartStax corn in which multiple modes of both herbicide-tolerance and insect-resistance are stacked (SmartStax). The functional unit was defined as "the production of 1 ton of corn at field gate in the state of Nebraska using either of the 4 described corn alternatives".

2.2. Input data and assumptions

Main input data are taken from the 2011 Nebraska Crop Budgets (Wright et al 2011). The Nebraska Crop Budgets are annually updated projections for crop cultivation produced by the Extension Division of the Institute of Agriculture and Natural Resources at the University of Nebraska - Lincoln in cooperation with the Counties and the United States Department of Agriculture. Budget 8 describes the cultivation projections for non-GM corn, budget 9 for the RR&Bt alternative and budget 10 describes the main input data for SmartStax. The input data for the RR alternative were interpolated from the non-GM alternative in relation to the use of insecticides and from the RR&Bt alternative in relation to herbicide treatments and the field operations of no-till cultivation practice (see Table 1).

Table 1. Amount of herbicide and insecticide active ingredient (a.i) applied for non-GM and 3 GMO corn varieties (kg a.i/ha).

	Active ingredient	**non-GM**	**RR**	**RR&Bt**	**SmartStax**
Herbicides	Atrazine	1,82	1,85	1,85	1,85
	S-Metolachlor	1,43	1,5	1,5	1,5
	Prosulfuron	0,0075	0	0	0
	Primisulfuron-methyl	0,0025	0	0	0
	Glyphosate	0	2,37	2,37	2,37
Insecticides	Fibronil	0,29	0,29	0,058	0,015
	Chlorpyrifos	0,084	0,084	0,017	0,0042
	Bifenthrin	0,018	0,018	0,009	0,009
	zeta-Cypermethrin	0,0028	0,0028	0,0028	0,0028

Yield for the RR alternative was modelled from the RR&Bt alternative based on a yield comparison of two transgenic herbicide-tolerant and insect-resistant varieties with their isogenic non-Bt counterparts (Haegele and Below 2013) showing that the Bt trait conferred an on average 11,8% yield benefit. As stated in the Crop Budgets, the alternatives are assumed to be in a continuous corn system (i.e. no crop rotation) and rainfed (i.e. no irrigation). In consultation with an expert panel from the University of Nebraska – Lincoln and in line with the recommendation in the Crop Budgets, the study assumed equal nitrogen use efficiency for all 4 corn production alternatives. Soil erosion is at 2,24 to/ha/year for the alternative using conventional tillage and 0,62 to/ha/year for the alternatives applying no-till practices (W. Vanek, USDA-NRCS Nebraska, pers. comm.).

2.3. Impact assessment

The environmental assessment is based on established life-cycle impact assessment (LCIA) categories as used also in Eco-Efficiency Analysis (Saling et al. 2002) and a wide range of other LCA approaches. Primary energy consumption, resource depletion, emission to air, emission to water and solid waste are assessed. The ecotoxicity potential of all chemicals intentionally released into the environment, e.g., fertilizers and pesticides is accounted for (Saling et al. 2005). Land use is assessed using to the scheme of the ecosystem damage potential (EDP; Köllner and Scholz 2007, Köllner and Scholz 2008). Consumptive water use is assessed as described by Pfister et al (Pfister et al. 2009).

Additionally, environmental indicators addressing the specific impacts of agricultural activity on biodiversity in agricultural areas, and on soil health and conservation, have been incorporated into the methodology (see Schoeneboom et al, 2011 and references therein). More specifically, biodiversity is assessed using the “driving force – state – response” model which is proposed by OECD to structure the complex relationships between agriculture and biodiversity (OECD 2003). The indicator set for biodiversity compiles factors which negatively affect biodiversity resulting in a decline of the state of biodiversity such as low crop rotation or high farming intensity. It also takes into account the effect of the applied pesticides by integrating ecotoxicological characterization factors from USEtox (Rosenbaum et al. 2008). The “state” indicator quantifies the status quo of biodiversity using a proxy such as the widely accepted bird indicator (Bird indicator 2013). “Response” indicators reflect activities which are able to promote or conserve biodiversity such as high crop rotation or the adoption of field margins or flower strips (for details see Saling et al. 2014).

The impact category of soil health is comprised of indicators assessing soil organic matter balance, soil erosion, soil compaction and the nutrient balances of the main nutrients N, P, K. More specifically, the depletion and build-up of soil organic matter is assessed as a balance derived from estimations by the IPCC LULUCF for cropland (IPCC LULUCF), or alternatively on the standard German cross-compliance ‘humus balance’ model (VDLUFA 2004). An assessment function, developed by Hülsbergen in 2003 (quoted in Ehrmann & Kleinhanss, 2008 and Christen 2009) is then used to evaluate the balance results. Nutrient balance is assassed as a function of the amount of fertilizer applied and the amount of nutrients retrieved through harvest. This balance is furthermore corrected for the ability of the soil to mineralize and thereby provide nutrients as indicated by soil nutrient supply classes. The nitrogen balance also considers different sources e.g. nitrogen fixation by leguminoses. The result of the nutrient balances is subsequently evaluated, using nutrients specific models with optimal scores around an equal nutrient balance of zero and decreasing scores for either nutrient deprivation or over-fertilisation. (Christen 2009). The potential for soil erosion is calculated default using the Universal Soil Loss Equation (USLE). This equation predicts the long term average annual rate of erosion on a field, based on slope, rainfall pattern, soil type, topography, crop system, and management practices (Hudson 1993).

In terms of economic assessment, both production costs as well as economic performance are taken into account. Production costs are grouped into variable and fixed costs and quantified using an overall total cost of ownership for the defined functional unit (Kicherer et al. 2007). Economic performance is assessed using indicators for farm profitability as the central criterion for economic sustainability, subsidies which may excert distorting economic effects and productivity as measured as the production value of agricultural goods per hectare weighted by the contribution of the agricultural sector to the national GDP.

The social assessment in AgBalance is derived from the SEEBALANCE method for social LCA, which was developed in 2005 by Universities of Karlsruhe and Jena, the Öko-Institut (Institute for Applied Ecology) Freiburg e.V., and BASF (Schmidt et al. 2005, Kölsch et al. 2008). Based on the UNEP-SETAC guidelines for social LCA of products 5 stakeholder categories were defined: Farmer, consumer, local community, internal community and future generations. The SEEBALANCE indicators and data sources are employed to assess the social impacts of industrial up- and downstream processes. For the agricultural activities in the life cycle, a set of adapted social impact indicators was integrated into the AgBalance method which were designed to match closely the same social sustainability topics addressed in the assessment of the upstream and downstream processes.

3. Results

3.1. Environmental impact assessment

The number of field operations as described in the Nebraska Crop Budgets are 14 in case of non-GM corn and 8 for the GM alternatives. The adoption of no-till practices makes the field operations of chiseling and chopping stalks obsolete and changed type of fertilizer used and its mode of application. These changes in field management reduced labor and machinery costs, but also significantly reduced fuel use for field operations from 62,78 l/ha for the non-GM alternative to 33,62 l/ha for SmartStax. Expressed in relation to the customer benefit of 1 ton of corn, fuel use for field operations is reduced by 60,4%.

The significantly reduced fuel consumption per functional unit is also reflected in better scores in the impact categories energy consumption and resource consumption. Both categories however are dominated by the energy and resource requirement for fertilizer production (see Figure 1 for energy consumption). In fact the production and use of fertilizer constitutes roughly 3 quarters or more of the impact in these categories.

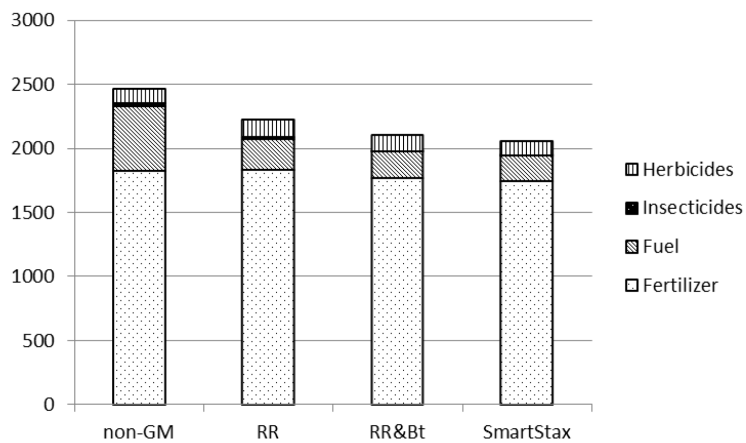

Figure 1: Total energy consumption derived from the production and use of the major inputs (in MJ/customer benefit of 1 ton of corn)

The footprint of all 8 environmental impact categories considered (Figure 2) shows furthermore the positive effect of higher productivity on land use - which is for over 95 % determined by land used for agriculture - and reduced by 26 % from non-GM to SmartStax due to higher yields.

The impact on the local biodiversity potential is determined among others by the ecotoxicology of the crop protection products used: whereas the herbicide tolerance trait requires higher use of herbicides, although with a more favorable eco-toxicological profile, the insect-tolerance in the alternative RR&Bt as well as SmartStax leads to a reduced intensity of use of insecticides with a potential positive impact on beneficial insects and thus a more favourable score for the local biodiversity potential. The other indicators in this impact category are equal (e.g. crop rotation, protected areas, agri-environmental schemes) or more or less similar (nitrogen surplus).

In the impact category soil the benefits of no-till on soil erosion are taken into account whereas the figures for soil compaction, soil organic matter balance as well as the nutrient balances are more or less equal for the alternatives considered. Soil erosion is decreased by 1,61 ton/ha/year or 72 % by the adoption of no-till cultivation practice. The impact of consumptive water use seems greatest of all environmental categories, but since the study assessed rainfed corn, differences in water use are related to the application of fertilizer and pesticides only. As a consequence the result of this impact category looks very favorable for the alternatives with insect-resistance, but in absolute terms the savings in water are of limited impact.

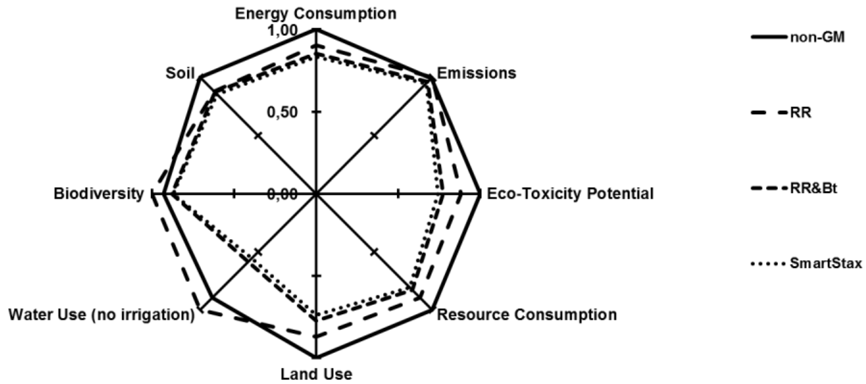

Figure 2. Representation of relative environmental impact category results for the different alternatives in fingerprint diagram. Relative improvement in each impact category is represented by smaller values on the respective axes; hence the smaller the fingerprint, the better the relative performance of the corresponding alternative.

For the impact assessment of emissions, both emissions to air and water as well as solid waste are considered. The final score is more or less unchanged in the sum of all aspects considered since this impact category is highly determined by emissions related to the production and use of nitrogen fertilizer. For example, greenhouse gas emissions are dominated by 89% in the case of the alternative non-GM to up to 95% in case of SmartStax by production of fertilizer and field emissions from nitrification processes in soils (data not shown). Since equal nitrogen use efficiency is assumed for the 4 alternatives the impact in this category are similar.

3.2. Socio-economic impact assessment

The economic assessment shows clear results both in terms of reduced fixed and variable associated with GMO corn alternatives as well as higher margins and farm profits (Table 2). Fixed costs related to machinery, buildings and equipment is reduced by up to 37% and variable costs by 22% between non-GM and SmartStax when expressed per functional unit. Profits per hectare have more than tripled.

Table 2. Overview of costs, revenue and profit.

	Unit	non-GM	RR	RR&Bt	SmartStax
Fixed cost	USD/1 t corn production	93,81	69,15	60,99	58,65
Variable cost	USD/1 t corn production	100,53	94,04	75,39	78,49
Total cost	USD/1 t corn production	194,34	163,19	136,38	137,14
Total cost	USD/ha	1036,80	993,82	941,69	989,98
Revenue	USD/ha	1280,40	1461,60	1657,20	1732,45
Profit	USD/ha	243,60	467,78	715,51	742,47

A main point of criticism for GMO corn is often the high price for seed. The results in Table 3 show that first of all, cost of seed are not the main variable cost, but rather fertilizer (see Table 3). And secondly, the higher cost for seed is more than compensated for by reduced cost for pesticides especially insecticides. For example, for SmartStax almost $ 9,97 extra seed costs per functional unit have to be paid, but this investment in return leads to reduced pesticide cost of $ 30,49 per functional unit. In other words, for every dollar spend on seeds, the farmer receives 3 dollars in return.

Table 3. Costs for main inputs ($/functional unit).

	non-GM	RR	RR&Bt	SmartStax
Seed	13,91	15,74	16,90	23,88
Fertilizer	25,83	28,23	27,59	27,31
Herbicides	13,68	11,58	10,21	9,77
Insecticides	25,22	22,10	4,24	1,36
Total	78,64	77,66	58,94	62,31

The assessment of the social impacts shows a more or less equal result for all alternatives in the sum of all impacts considered. For example, positive effects of higher productivity on the statistical occurrence of working accidents or occupational diseases are balanced by reduced employment opportunities and reduced contributions to social security funds as expressed per functional unit (data not shown). All other social impact indicators remain unchanged since the study investigates different alternatives for same crop in same region and same year.

3.3. Scenario analysis

In total 4 different scenarios are calculated to investigate the robustness of the model in relation to major assumptions made in the study as well as major trends in corn cultivation. First of all, the study calculates with a lower yield potential for non-GM corn as based on the input data from the Nebraska Crop Budgets. Whereas the yield advantage of insect-resistant corn varieties seems well established, a clear effect on yield by the herbicide-tolerance trait is more difficult to establish and seems to depend more on the context of cultivation (Shi et al. 2013). Scenario 1 therefor calculates with a yield level for non-GM similar to RR corn. As a result, the environmental impact of non-GM corn is now similar in the environmental impact categories emissions, eco-toxicity potential and land-use. Benefits for RR remain in relation to resource and energy consumption (related to reduced diesel use) and soil erosion whereas biodiversity and water consumption is unaltered (Figure 3). In the sum of all environmental and socio-economic impacts considered, there is no significant difference between non-GM and RR in this scenario whereas the performance of alternatives RR&Bt and SmartStax remains significantly better.

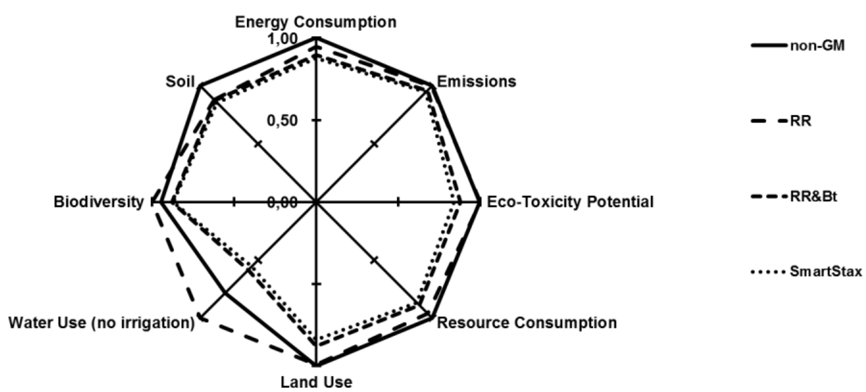

Figure 3. Representation of relative environmental impact category results for the different alternatives in scenario 1 in fingerprint diagram. Relative improvement in each impact category is represented by smaller values on the respective axes; hence the smaller the fingerprint, the better the relative performance of the corresponding alternative.

Evolution of resistance in weeds and pests to the herbicide and insecticides applied can reduce the efficacy of crop protection strategy. This is by no means restricted to the use of GMOs but is known in all areas where crop protection products are used intensively (Powles et al, 1997). Glyphosate-tolerant weeds have increasingly been reported in the US since the 2004/2005 growing seasons (Heap 2014). In recent years Bt–resistant pests (Tabashnik et al. 2013) have been reported that threaten the efficacy of Bt crops. In scenario 2 and 3, a 50% increase in the use of herbicides or insecticides, respectively, are assumed to model the potential impact of the consequential higher rates of pesticide use. The results of these scenarios show that the environmental impact of higher rates of pesticides is very limited since most environmental impact factors are dominated by fertilizer production and use as well as productivity (data not shown).

Finally, scenario 4 assessed the impact of changing from continuous corn cultivation to a system where corn follows soybean cultivation. High demand for corn as feed and for biofuel production has shifted crop rotation patterns from corn-soy rotations to continuous corn over the last decade (Plourde et al. 2013). A yield penalty has been associated with long-term continuous corn cultivation and nitrogen availability identified as a major factor (Gentry et al. 2013). The Nebraska Crop Budgets 2011 specifically include a budget for this scenario (budget 11) which assumes a 33% reduction in use of inorganic nitrogen fertilizer due to the availability of nitrogen fixed by the soybean nodule mycorhizza and an increased yield potential from 110 bu/acre to 115 bu/acre. The Crop Budgets assume no change in pesticides applied for this scenario. These assumptions were modelled on alternative RR&Bt in scenario 4. As a result, the impact of fertilizer production was greatly reduced in the categories greenhouse gas emissions, energy and resource consumption (Figure 4). Also local biodiversity potential improved through better crop rotation. The overall environmental score has improved significantly whereas the socio-economic score is highly dependent on the revenue prices for corn in relation to soybean (data not shown).

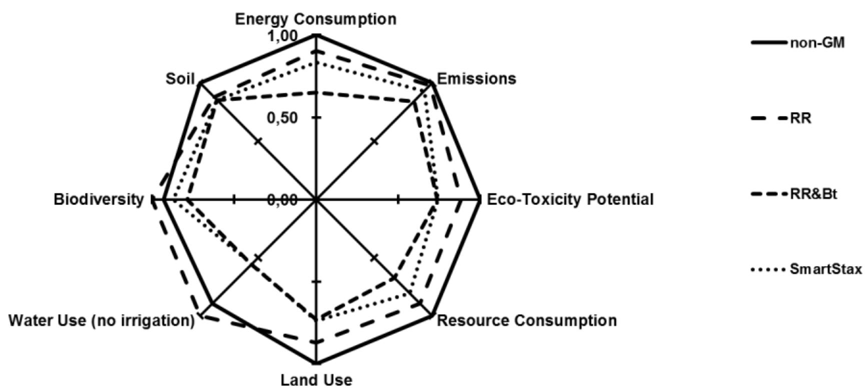

Figure 4. Representation of relative environmental impact category results for the different alternatives in scenario 4 in fingerprint diagram. Relative improvement in each impact category is represented by smaller values on the respective axes; hence the smaller the fingerprint, the better the relative performance of the corresponding alternative.

4. Discussion

Since their introduction in 1996, GMO traits have become standard technology incorporated into the major row crops in the Americas as well as cotton production globally. This rapid adoption makes clear that there are agronomic and economic benefits for the farmer from use of GMOs. This stands in contrast to the public perception that exists around GMOs. The GMO traits that are available today do not offer a direct customer benefit and this may have hampered acceptance of the technology. It is therefore even more important to assesses the potential impacts of GMO traits in a holistic and comprehensive way and use the results to inform the public and as guidance for further development. We used an LCA-based methodology assessing environmental, economic as well as social impacts of GMO traits in this study.

We could show that there are significant advantages associated with the use of GMO insect-resistance traits. It is accepted that Bt traits are responsible for higher and more stable yield levels (Shi et al., 2013). This yield advantage alone is responsible for a largely more positive ecological footprint. Reduction in use of insecticides furthermore has positive effects on biodiversity since less non-selective products are sprayed that harm both pest and beneficial insects. Insecticides are also a major variable cost and the investment in insect-resistant traits certainly pays-off for the farmer and it is easy to understand these GMO have become popular where Lepidopteran pests are common. The rising occurrence of insects resistant to Bt is now threatening the benefits of this trait. We could show in a scenario analysis that a 50% increased use of insecticides does not translate into a higher overall environmental burden. However, when Bt resistant insects can no longer be controlled effectively by increasing amounts of pesticides and negative effects on yield levels are occurring, this would directly negatively affect the environmental footprint as well as the economic performance of these traits. Appropriate resistance management programs need to be adopted by farmers to avoid any further development of resistant pest.

The impact of the herbicide-tolerance GMO trait is more ambivalent. A positive yield effect of herbicide-tolerant crops is more difficult to establish and seems to depend on type of herbicide-tolerance as well as geographical context (Shi et al., 2013). The scenario analysis assuming similar yield between non-GM and RR shows that without a yield benefit, the sustainability impacts of both systems are very much comparable. In contrast to earlier reports (see for example Brookes and Barfoot 2013), this study suggests herbicide-tolerant GMO corn is associated with higher levels of herbicide application. It is important to note however that the (eco-)toxicological profile of glyphosate, the herbicide that is most commonly used in combination with a herbicide-tolerant GMO corn, is more favorable than most other herbicides. As a result, even in a scenario where 50% higher application of glyphosate is assumed, the overall impact of herbicides on the environmental footprint of corn cultivation is low. Again, and analogous to Bt resistant pests discussed above, when herbicide-tolerant weeds can no longer been controlled effectively and negative impacts on yields are occurring, this most certainly will negatively impact the sustainability of this GMO trait.

Although no-till systems are adopted in regions and crops where GMO traits are not prevailing, it is generally acknowledged that herbicide-tolerant crop varieties have favored the adoption of no-till in soybean and corn in South and North America (Brookes and Barfoot, 2013). The advantages in soil conservation – i.e. preventing soil erosion and to a lesser extend enabling build-up of organic matter – are well described and are an important factor for long-term farming sustainability. Also in this study, soil erosion is reduced by 1,61 ton/ha/year or 72 % by the adoption of no-till cultivation practices. No-till is also the main driver for the reduction by over 60% in fuel use for field operations. This study has not accounted for a beneficial effect on greenhouse gas emissions through carbon sequestration in soil and this remains an upside potential for no-tillage cultivation systems that remains to be explored.

Whereas the impact from the production and use of pesticides was shown to be limited, this study clearly demonstrated that the highest ecological impact in corn cultivation is associated with the production and use of inorganic fertilizer. The production of the fertilizer requires large quantities of natural gas and accounts for roughly 75 % of all energy and resources consumed. Concurrently, emissions from production facilities together with emissions from its use in the field due to nitrification processes in the soil accounts for up to 95% of greenhouse gas emissions related to corn production. From these figures it becomes clear that any strategy to reduce inorganic fertilizer input for example by genetically increasing the nitrogen use efficiency of the crop (Xu et al 2012) or reducing the nitrification processes in soil (Liu et al 2013) will benefit the sustainability of corn production substantially.

The high impact of fertilizer production is also illustrated in a scenario looking at potential impacts of a corn-soybean rotation instead of continuous corn cultivation. A 33% reduction in inorganic nitrogenous fertilizer directly translates into significant reductions in air emission as well as energy and resource consumption. In addition, this crop rotation has a favourable impact on biodiversity and could be integrated into resistant management practices needed to maintain efficacy insect-resistance traits. These environmental benefits of a soy-corn rotation should be investigated further in a comprehensive and consequential assessment of both elements in crop rotation together with a proper economic assessment.

Next to fertilizer, the yield level of the different alternatives is of course a major factor in any assessment which uses a productivity factor as functional unit such as “a ton of corn produced” in this study. The yield level is a product of the genetic potential of the crop, the environment it is grown in and the cultivation practice used and these factors are highly interdependent. As such the yield levels of the alternatives in this study are a product of continued breeding efforts, incorporation of traits through biotechnology and changes in management practices. From reports of side-by-side comparison of GMO varieties with their non-GM isogenic counterparts a positive effect on yield is demonstrated for the insect-resistant trait (Haegele and Below 2013). Likewise other genetic approaches to increase the intrinsic yield potential (Condon et al 2002) or enhance resilience of crops against pest, diseases or stresses have great potential to benefit the sustainability of corn production.

5. Conclusion

The study shows that GMO traits contribute to more sustainable corn production in Nebraska. The positive impact of the Bt insect-resistance trait is associated mainly with higher yield levels or less yield risk, reduced cost and to a lesser extent reduced environmental burden from insecticide use. Glyphosate herbicide-tolerance can have a positive impact too if the trait contributes to higher yields - or at least not less yield - and leads to a higher adoption of no-till cultivation practice which in turn is favorable for soil erosion and leads to reduction in field operations and associated fuel use.

Overall the study shows that the environmental impact of the production and use of pesticides is low and that highest burden is from the production and use of inorganic fertilizer. Yield is another main driver for the sustainability of corn production. Likewise, crop varieties, farming practices or technologies that reduce the need for inorganic fertilizer, increase the yield potential or its resilience against pest, diseases and stresses should be the main targets for sustainable development in corn production..

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Comparing two LCA approaches for the transport of milk from farms to processing plants in Switzerland

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ABSTRACT

The environmental impact of transporting milk from farms to processing plants depends mainly on the distance and the operating parameters of the vehicles (Foster et al. 2006). The aim of this study is to compare different LCA approaches for the quantification of the environmental impact of a milk collection round trip in Switzerland. We considered two approaches: The simplified approach, where the average distance between farms and processing plant and standard emission factors for the relevant truck type were used and a detailed approach, where the effectively driven round trip was considered. For all analyzed impact categories, the simplified approach results in a smaller impact compared to the detailed approach. The selection of a modeling approach for transports depends on the way how a transport is being organized but also on the importance of transport's impact of the whole life cycle.

Keywords: LCA, transport, milk, collection round trip

1. Introduction

In Europe, food and drink products are responsible for 20-30 % of the environmental impacts of private consumption (Tukker et al., 2006). Although transport occurs between each step of supply chains, it often has an insignificant influence on the total environmental impact of agricultural products, especially of animal products such as milk. Bystricky et al. (2014) conclude after analyzing five different agricultural products (plant and animal products) from different provenance that the part of the total environmental impact of transportation is larger for products with small environmental impact per unit such as potatoes. Foster et al. (2006) give a value of 0.0068 kg CO_2 for direct emissions per kilogram milk transported, which corresponds to 0.66 % of the total greenhouse gas (GHG) emissions from milk at farm gate. Due to this negligible role of transports on the total environmental impact of food products, transports are often modelled with average parameters in LCA. For example, Pattara et al. (2012) calculated the environmental impact of wine transportation using averaged values. Further examples can be found in the publications of González-Garcia et al. (2013), Heller et al. 2008 and Browne et al. (2005). In all those studies, transports were considered based on average transport distances between point of collection and point of delivery and average life cycle inventories. Average life cycle inventories for transport, such as those in ecoinvent, are based on average load and empty trip factors. In the case of ecoinvent, the load factor including empty trips for lorry transports used is 50 % (Spielmann et al. 2007). However, the variability between single transport services is very high, thus such average factors might not be representative. In the case of line haul transports for example, load factors can be as high as 100 % with no empty trips. In this situation a fully loaded truck will only drive the distance once to transport its products, in a situation where the truck would have been only half loaded, it would have to drive the distance twice to transport the same amount. Considering a higher fuel consumption of fully loaded trucks of approx. 21 %, which is the difference in fuel consumption of the two processes "operation, lorry 20-28t, fleet average" and "operation, lorry 20-28t, full, fleet average" (both from Spielmann et al. 2007), total fuel consumption for a transport in a fully loaded truck would still be about 40 % lower compared to a transport in a half loaded truck. Thus, using average emission factors for line haul transports would lead to an overestimation of the environmental impact. On the other hand, for transports with lower load and higher empty trip factors, averaged data would lead to an underestimation. This might be the case for transports organized as collection or delivery round trips, which are distinguished from common transport services by the European standard EN16258:2012 – "Methodology for calculation and declaration of energy consumption and GHG emissions of transport services (freight and passengers)".

An agricultural product that is typically transported in collection round trips is milk. The goal of this study is to find out whether there is a significant difference between a detailed LCA of the collection round trip and a simplified approach using average transport inventories and average direct distances from the points of collection to the point of delivery.

2. Methods

Two different LCA approaches for the quantification of the environmental impact of a milk collection round trip in Switzerland are compared. One approach is a simplified approach, where standard life cycle inventories for the relevant truck type and average distances between farms and dairy processing plants were used. The second approach is more detailed, as the effectively driven distance and specific fuel consumption are considered following the recommendations for collection round trips of the European standard EN16258:2012 (CEN, 2012). The calculations are performed on one exemplary milk collection round trip in the canton of Thurgau in Switzerland. A carrier specialized in milk transports provided the primary data, background data were taken from the ecoinvent V2.2 database (ecoinvent, 2010).

2.1. Functional unit and system boundaries

The functional unit is the delivery of one kilogram milk. The system boundaries comprise all transport related life cycle phases such as direct emissions, fuel production, vehicle manufacturing, maintenance and disposal, and road infrastructure. The milk production, processing, trade and consumption are excluded in this study (Figure 1). The time boundary is the year 2013.

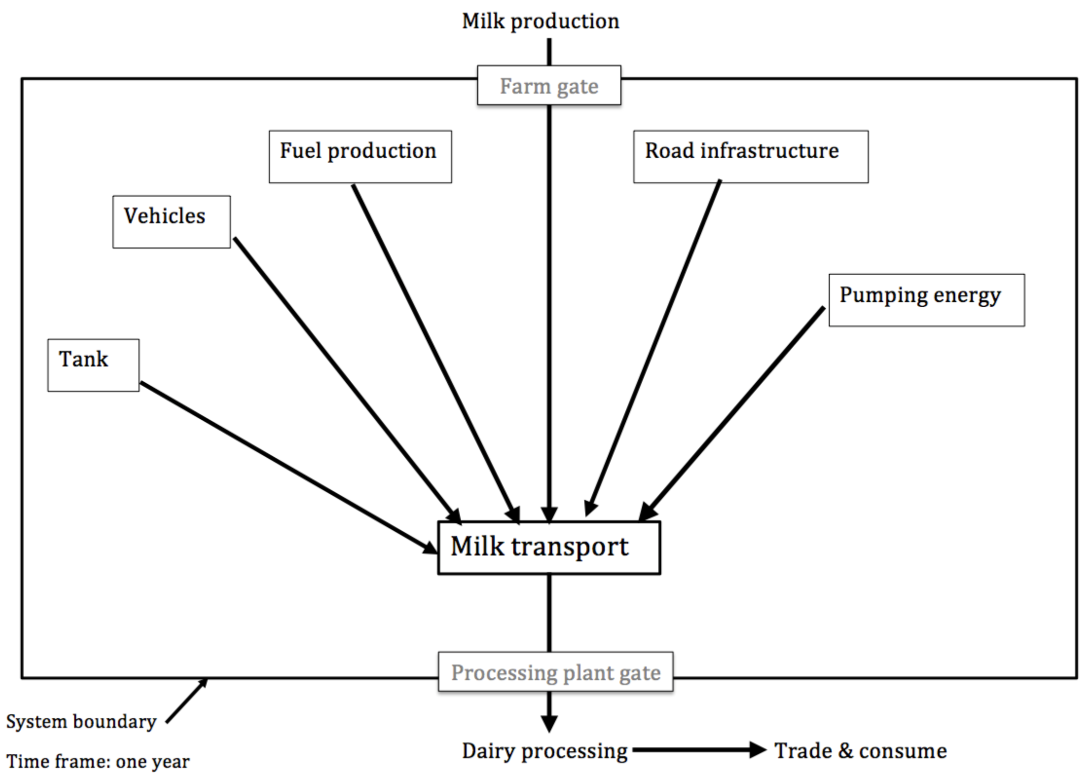

Figure 1. Initial system boundaries definition.

2.2. Impact categories

The European standard EN16258:2012 (CEN 2012) is a guideline for the calculation of GHG emissions and energy consumption of transport services. Therefore we decided to use those two impact categories for our study, assessing the global warming potential (GWP) as well as the non-renewable energy demand (NRED) (fossil and nuclear energy). These two categories are thought to be the most important categories for environmental impact quantification of transport services since they are directly influenced by the fuel combustion process and production, respectively. In order to complete the picture, we decided to integrate the impact category human toxicity (HT), as it was the impact category contributing the third-most to the aggregated environmental impact when using the impact assessment method ReCiPe 2008 (H) with normalization for Europe. Table 1 shows the considered impact categories and the applied methods.

Table 1. Impact categories and corresponding impacts assessment method and impact factor's unit.

Impact category	Impacts assessment method	Impact factor's unit
Non-renewable energy demand (NRED)	According to ecoinvent; Frischknecht et al. 1998	MJ eq
Global warming potential (GWP)	IPCC 2007, 100a	kg CO_2e
Human toxicity (HT)	USES-LCA 2.0	kg 1,4-DB eq

2.3. Data collection

2.3.1. Selection of the round trip

A milk collection round trip in the canton of Thurgau in the northeast of Switzerland was selected. As only one round trip is analyzed, and the variability of milk collection round trips are probably high, results for this route are not expected to be representative. However, it will allow us to highlight some trends in the environmental impact calculations of transport.

The analyzed round trip is done by truck and runs every second day of the year. It collects milk from 21 farms and is approximately 80 km long. The driven distance includes empty runs. At the farms, a pump powered by the truck's engine pumps the milk into the truck's tank which means that the truck engine never stops running during the whole collection round trip (oral communication).

2.3.2. Data sources

The carrier provided primary data for the round trip, truck type and fuel consumption. The tank vendor provided data for the tank. All data was collected for the calendar year 2013. The provided fuel consumption data was not measured, but estimated by the carrier. They were 30 % higher compared to the average fuel consumption of the same truck type from the corresponding ecoinvent process, which can be explained by the extra diesel consumption for the pumps, and therefore we considered them as plausible. Data for road and vehicle infrastructure were based on ecoinvent v2.2 and literature. Demand factors were calculated following the same procedure as in ecoinvent (Spielmann and Scholz, 2005).

2.3.2. The two modelling approaches

The simplified approach is calculated with the ecoinvent transport process "transport, lorry 20-28t, fleet average", a truck type comparable to the one used for the studied transport service. This process includes all the above mentioned transport related life cycle phases for transports with average load factors. The functional unit of this process is a ton kilometer. The ton kilometers needed for the delivery of one kilogram milk are calculated taking the average of the distances from each farm on the collection round trip to the processing plant.

The detailed approach is based on the same ecoinvent process, with some adaptations to reflect the real situation in a better way. First, instead of using ton kilometers, the process "transport, lorry 20-28t, fleet average" is converted from ton kilometers to vehicle kilometers, which allowed us to consider the whole collection round trip, starting and ending at the parking place of the truck according to the definition of vehicle operation systems (VOS) in the European standard EN16258:2012 (CEN 2012). The truck manufacturing, maintenance and disposal is changed according to the effective vehicle weight and complemented with the tank. For the operation of the truck, fuel consumption on the round trip is adapted, so that it includes the required energy for the pumping of the milk into the tank at the farms and collecting sites. Afterwards the total emissions of this trip are divided by the amount of milk delivered per trip, resulting in emissions for the delivery of one kilogram milk.

3. Results

Transport life cycle was divided into eleven phases for the detailed approach: 1) truck manufacturing; 2) truck maintenance; 3) road operation, maintenance; 4) road construction; 5) truck tank; 6) trailer tank; 7) truck operation; 8) truck disposal; 9) road disposal; 10) truck tank disposal; 11) trailer tank disposal. The results are shown in Figure 2.

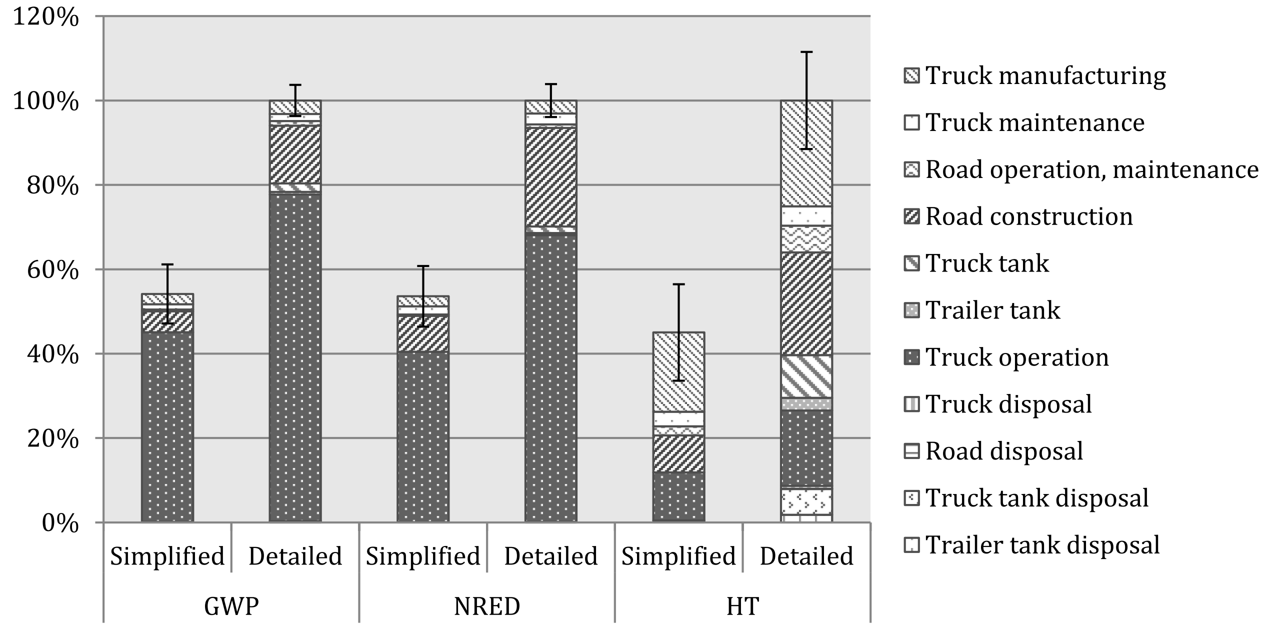

Figure 2. Comparison of the simplified and detailed approaches for the impact categories GWP, NRED and HT.

For the categories global warming potential (GWP) and non-renewable energy demand (NRED), the impacts of the simplified approach are approx. 50 % lower than those of the detailed approach and for the category human toxicity (HT) approx. 55 % lower.

In our detailed approach, the total GWP is 0.00517 kg CO_2e / kg delivered milk. The truck operation contributes 77 %, due to the direct emissions of the fuel combustion and from the fuel production linked emissions. The rest of the CO_2e emissions are caused mainly by the road construction which contributes 14 % to the GWP.

Total NRED in the detailed approach is 0.08170 MJ eq / kg delivered milk. Like for GWP, the truck operation contributes the most with 68 % of total NRED. This apparent correlation between the category GWP and NRED arises from the fuel consumption. NRED impact is determined by the fossil energy use. The impact on GWP is determined mainly by the combustion of the fuel for transportation.

For HT, the transport cause 0.00068 kg 1,4-DB eq / kg delivered milk. This impact is, compared to GWP and NRED, caused by other sources: 25 % of the total impact is caused by truck manufacturing and further 24 % by road construction. During the manufacturing of the truck reinforcing steel dust is generated and needs to be disposed. The impact on HT of the road construction arises from the bitumen which is produced with crude oil. During the crude oil extraction, formation water needs to be drained. Further, two other phases influence this impact category: truck tank (10 %) and truck operation (18 %). Same as for the road construction, the impact of the truck operation arises from the fuel production which requires crude oil extraction. Moreover, the impact on HT of the truck tank is due mainly to the chrome steel production which requires the burning of bituminous coal as energy source. This source of energy emits a lot of particles to the air (mining and burning) and, in addition, ashes emerge which need to be disposed in a residue landfill.

4. Discussion

4.1. Most important sources for environmental impacts of transports

Considering the main contributors to transport emissions, the results of this study are in line with the results of other LCA studies on transportation. The main cause for environmental impacts of transport arises from the fuel production and combustion for the impact categories GWP and NRED. These results correspond to the findings of Spielmann and Scholz (2005), who affirm that the operation component has the largest environmental impact of transport services. Eriksson et al. (1996) also determined the fuel combustion as having the largest impact followed by the fuel production.

For HT, Spielmann and Scholz (2005) found that the particulate matter 2.5 (PM2.5) and non-methane hydrocarbons (NMHC) emissions are influenced mainly by the transport infrastructure. PM2.5 can arise to some extent from the bituminous coal extraction and burning (WHO 1986). In our studied transport, the main causes for HT are also related to transport infrastructure, such as roads and vehicles.

4.2. Putting it into the larger context – the impact of transport in relation to the whole life cycle of products

According to the results of Browne et al. (2005), who reviewed different studies concerning supply chain LCA, the proportion of energy accounted for by commercial freight transport varies in many cases from 2 to 9 %. In a study on the carbon footprint of milk in Switzerland, Alig et al. (2011) calculate an impact of 5 MJ eq per kilogram milk at farm gate. The value for the category NRED obtained in the present work for milk collection transportation represents 1.6 % of this value. For the whole life cycle of milk, the importance of the energy consumption of milk collection will be even lower, proving that the transport life cycle phase is almost not relevant for animal products. This was also observed by Bystricky et al. (2014).

For GWP, Alig et al. (2011) found emissions of 1.3 to 1.4 kg CO_2e per kilogram milk at farm gate, thus our 0.00517 kg CO_2e per kilogram delivered milk only result in 0.4 % of this impact, which is a very small share. Our transport related values for GWP per kg of delivered milk are slightly lower than found in other studies. Foster et al. (2006) give a value of 0.0068 kg CO_2 for direct emissions per kilogram milk transported. Since the procedure and assumptions for the modeling of transportation are not documented in the study of Foster et al. (2006), it is difficult to assess the differences in results. One explanation could be that the study of Foster et al. (2006) was conducted in Great Britain and that the transport infrastructure and distances differ there from the ones in Switzerland. In our analyzed collection tour, all farms situated within a small perimeter from the processing plant. Another difference could arise from the used transport inventories and emission factors.

Like other studies already pointed out (Eide 2002; Flysjö et al. 2011; Foster et al. 2006; IDF 2009), the transportation impact on GWP and on NRED in the milk production life cycle is very low. The farming phase is much more important with about 75 to 95 % of the total environmental impact of milk production (Berlin 2002; Flysjö et al. 2011; Foster et al. 2006; IDF 2009).

4.3. Why does transport still matter?

Most literature studies use average direct distances for the modeling of transport in their calculations. For example, Pattara et al. (2012) calculated the environmental impact of wine transportation based on “the number of bottles delivered, average distance travelled from the firm to the final market, type of vehicle used and its loading capacity”. Further examples are to be found in the publications of González-Garcia et al. (2013), Heller et al. (2008) and Browne et al. (2005) as average transport distances, average datasets and distance between the point of collection and delivery are considered respectively. However, if transports are organized as collection round trips, using average direct distances from the collecting points to delivery point and average emission factors might lead to an underestimation of transport’s impact. If such a transport is modelled with the effective driven distance, the environmental impacts on the categories GWP, NRED and HT are significantly higher, in our case by a factor of two. Even if in the case of milk transports are still rather insignificant, for products, where transports do matter, such as some plant products and products that are transported over long distances, it might be worth to have a closer look at transport processes. Average transport processes are what they are – just average. Real transport impacts could be both, much higher or lower.

5. Conclusion

In the supply chain of agricultural goods, transportation occurs many times and in different ways. In the present study, transportation of milk from farm to processing plant has been studied on a typical collection round trip in Switzerland. It could be shown that there is a significant difference in the results on transportation environmental impact between a simplified approach based on average distances and standard emission factors and a detailed approach based on the effective driven distance and specific emission factors. In all three analyzed impact categories GWP, NRED and HT, the simplified approach resulted in approximation half as much impact compared to the detailed approach. These differences arise mainly from the fact that the effectively driven distance of round trips are higher, leading to more use of road infrastructure and higher fuel quantity. At the moment, most LCA studies on agricultural goods including transportation in their scope do mostly calculate transportation with data based on average direct distances from collection to delivery points and use standard emission factors. Depending on how those transports are organized, this might lead to an under- or overestimation of transport's impact, as effectively driven distances or vehicle load factors might differ from those of average transports. In our example of milk transport, the impact of transport compared to the one of milk production itself was rather small, and even a doubling of transport's impact did not make it a major contributor to the impact of the whole milk life cycle. However, in cases where goods are transported over longer distances or for plant products that often have a lower environmental impact than animal products, we recommend to have a closer look at how transportation is calculated.

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Assessing GHG mitigation options for crops at regional level using ecosystem modelling and LCA

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ABSTRACT

Soil, climate and management practices make greenhouse gas (GHG) emission estimates associated with crop production highly uncertain. Biophysical modelling provides reliable reactive nitrogen (Nr) estimates for environmental assessment. In this paper LCA and ecosystem modelling are combined to improve GHG estimation from cropping systems in the Paris (France) area, and to compare environmental impacts of two cropping systems at regional level. A cropping system aimed at productivity with high environmental performance (PHEP), while the other one aimed to reduce (GHG) emissions by half (50%GHG). Model-derived GHG estimates for crop production were at least 7% lower than estimates from the standard methodology applied to LCA, emphasizing the importance of regional factors in agricultural LCAs. The 50%GHG cropping system appears promising (184% reduction in the life-cycle GHG emissions) for climate mitigation of arable crops, pending trade-offs with other impact categories.

Keywords: LCA, GHG, regional level, CERES-EGC model, cropping systems.

1. Introduction

Modern agriculture contributes a large share of the anthropogenic emission of greenhouse gases (GHGs), in particular in the form of nitrous oxide (N_2O), a potent GHG driven by the application of fertilizer nitrogen (N) inputs (Saggar 2010). Agriculture can be a sink for CO_2 through soil C sequestration, by fixing carbon from the atmosphere and storing it on the short term with residues and soil organic matter (Brady and Weil 2002). Life-cycle assessment provides a relevant means of balancing these effects to estimate the carbon footprint of agriculture, and exploring mitigation strategies. However it has to date mostly been applied on a crop-specific basis (Brentrup et al. 2004a; Charles et al. 2006; JRC et al. 2007), whereas environmental impacts are better assessed with a cropping systems approach taking into account the interactions between successive crops and their environment (Goglio et al. 2014; Goglio et al. 2012). Also, most of the previous work on the LCA of crops involved short-term durations (one to a few years), too short to capture variations of soil organic carbon (SOC) stocks, and relied on simple models to estimate N_2O emissions (Brentrup et al. 2004b; MacWilliam et al. 2014; Nemecek et al. 2011a; Nemecek et al. 2011b). Since soil carbon and nitrogen (N) dynamics are highly influenced by farm management such as tillage, fertilizer application, residues management and their interaction with soil and climate (Brady and Weil 2002), local conditions should be considered to improve the estimation of GHG emissions. Direct measurements, especially for N_2O and NO, are often costly and time consuming, while the IPCC methodology, commonly used for N_2O estimations, does not account for local conditions. Agroecosystem models on the other hand are sensitive to these factors but require large amount of data (Gabrielle and Gagnaire 2008).

Despite this advantage in principle, only a few studies have integrated LCA with agroecosystem modelling to estimate GHG and other impacts during long term cultivation (Adler et al. 2007; Gabrielle and Gagnaire 2008; Goglio et al. 2014; JRC et al. 2007). However, all these studies were based outside Europe or involved a limited set of crops; for example, JRC et al. (2007) evaluated only biofuels crops for one year. Thus, the aim of the present work is to evaluate field emissions of CO_2 and reactive N (Nr) from contrasted cropping systems using the agro-ecosystem model CERES-EGC for the Ile de France region. The cropping systems were specifically designed with an *ex ante* evaluation to estimate the possible reduction in soil borne emissions using specific crop management. CERES-EGC model was run to simulate Nr emissions (as N_2O, NO, NH_3 and nitrate), along with soil C variations for 20 years. These fluxes were included in a cradle-to-farm-gate LCA of the biomass produced by the two cropping systems.

2. Methods

2.1. Cropping systems

The two cropping systems assessed over the Ile the France region were established in the ICC (Innovative cropping systems with constraints) trial in Grignon (40 km west of Paris) on a silty loam soil with the following aims: (1) to attain altogether High Environmental Performances and Productivity for the reference system (PHEP), and (2) to reduce by half the GHG emissions from the 50%GHG system compared to the PHEP system, while achieving the same environmental criteria (other than GHG emissions) as the PHEP system (Colnenne David et al. 2011).

The PHEP cropping system involved the following crop rotation: faba bean (*Vicia faba* var *minor* (Harz) Beck), winter wheat (*Triticum aestivum* L.), rapeseed (*Brassica napus* L.), winter wheat (*Triticum aestivum* L.), white mustard (*Sinapis arvensis* L.) or black mustard *(Brassica nigra* L.) as catch crop and spring barley (*Hordeum vulgare* L.). Main crop management characteristics of the PHEP system are given in Table 1.

Table 1. Selected characteristics of crop management for the cropping systems analysed on the Ile the France region (PHEP, 50%GHG), crop yields recorded during the 2009-2012 seasons.

System	Crop	Tillage	N fertiliser application[a] (kg ha^{-1})	Yield (Mg of dry grain ha^{-1})
50% GHG	**Spring Faba bean**	No tillage		0.63
	Rapeseed	No tillage	N 10	3.66
	Winter wheat	No tillage	N 70+43	7.52
	Winter Barley	No tillage	N 70+49	5.64
	Maize	No tillage	N 60+45	8.12
	Triticale	No tillage	N 72	6.48
PHEP	**Winter Faba bean**	Minimum tillage		1.37
	Winter wheat	Minimum tillage	N 50+37	7.34
	Rapeseed	Minimum tillage	N 50+60	3.74
	Winter wheat	Minimum tillage	N 70+58	7.34
	Spring Barley	Ploughing	N 65	5.70

[a]granular ammonium nitrate with no retarders

The 50%GHG cropping system maximizes the accumulation of soil organic carbon with the use of high biomass yielding cereals and by increasing the rate of crop residue return to soils. This is achieved with no tillage. Nitrous oxide emissions are mitigated by reducing mineral nitrogen fertilizer inputs with the introduction of legumes both as main crops in the rotation and as cover crops. Cover crops were also introduced to decrease the accumulation of soil nitrate and the subsequent emissions of N_2O from nitrate denitrification. Rapeseed was introduced to reduce nitrate leaching and the ensuing emissions of N_2O. The rotation is: (cover crop) faba bean, rapeseed, (cover crop) winter wheat, (cover crop) winter barley, (cover crop) maize (*Zea mays* L.), triticale (*XTriticosecale* (Camus) Wittm.). Main crop management details are given in Table 1 (Colnenne David et al. 2011).

2.2. Regional modelling

Crop yields, soil C dynamics and emissions of reactive N (Nr), including N_2O in particular were simulated with the agro-ecosystem model CERES-EGC over the Ile de France region, following the methodology of Gabrielle et al. (2014). The region is a 150 km x 150 km square area surrounding Paris, with 55% cropland. A GIS database was constructed with available geo-referenced data on this region, including administrative borders, land-cover type, crop management practices, soil properties and climate. The corresponding layers of spatial information were mostly in vector format, and overlaid to delineate elementary spatial units representing unique combinations of soil types, weather data, and agricultural management. These units were subsequently used in the CERES-EGC simulations at the field-scale, in a bottom-up approach to map the emissions. We used weather data predicted for the 2010-2030 time slice by the DRIAS project in France (Lémond et al. 2011), using the IPSL-CM4 model with the A1B GHG emission scenario from IPCC, which appeared as an intermediate scenario for air temperature and rainfall among the range of models and forcings tested by this project. Compared to the 1961-1990 period, air temperature would rise by 1.35 °C, with an average of 11.7 °C for the 2010-2031 time slice, and rainfall would remain constant at an average of 641 mm yr^{-1}. CERES-EGC was run for the current land-use (2010).

2.3. Life-cycle assessment

LCA was performed following a cradle to farm gate approach (Goglio et al. 2014; Goglio et al. 2012), considering grains as final products. System boundaries encompassed all agricultural inputs and farm machinery production from raw material extraction to transport to farm (Brentrup et al. 2004b). LCA was carried out using one ha of land and one GJ of grain energy output as functional units, according to previous research (Nemecek et al. 2011a; Nemecek et al. 2011b).

The impact categories evaluated were cumulative energy demand, global warming potential (GWP) with a 100 year horizon, acidification potential, and eutrophication potential, using the CML 2001 method integrated in SimaPro 7.3 (2012) (Guinée et al. 2001; Nemecek et al. 2011a; Nemecek et al. 2011b; SimaPro 7.3 2012). Toxicity impacts were evaluated with the EDIP 2003 method available in SIMAPRO 7.3 (Nemecek et al. 2011a; Nemecek et al. 2011b; SimaPro 7.3 2012), involving the following categories: human ecotoxicity for water, soil and air; chronic ecotoxicity for soil and water; acute ecotoxicity for water. The 5% and 95% percentiles of the modelled field emissions were used to evaluate the effect of spatial and temporal variability of emissions on LCA indicators. This procedure was adopted for all the reactive N species considered in this study (i.e. nitrous oxide, nitric oxide (NO), ammonia (NH_3) volatilisation and nitrate (NO_3^-) leaching) and also for soil CO_2 exchanges.

Within the system boundaries, farm transport from the field to the farm centre was the only post-harvest process considered. During cultivation, transport of farm machinery from the farm centre to the field and its return journey was also accounted for (Brentrup et al. 2004b; Gasol et al. 2012). No drying process, except hay drying was included in the present system according to local conditions, since all the grain is directly sold to the wholesale company at field moisture.

The life cycle elaboration was carried out with the SIMAPRO software with different data sources: *ex ante* evaluation data from the ICC trials, databases integrated in the SIMAPRO software (SimaPro 7.3 2012), data taken from literature (Audsley et al. 1997; Brentrup et al. 2004a; Brentrup et al. 2004b), CERES-EGC model results for soil GHG emissions in agreement with the ISO standards 14040 and 14044 (ISO 2006a; ISO 2006b) (Table 2).

Table 2. Main data sources for different processes included in the analysis of crop management for the cropping systems analysed in the Ile the France region (PHEP, 50%GHG)

Upstream processes for agriculture	**Technical operation during cultivation**	**Fuel and material consumption for each field operation**	**Soil GHG emissions and reactive N species**	**Pesticide fate emission**	**Transport processes**
Elaborated using databases integrated in SIMAPRO together with data taken from literature (Audsley et al. 1997; Brentrup et al. 2004b; SimaPro 7.3 2012)	*Ex ante* evaluation of the ICC trial	Elaborated on the basis of the tractor power (ASABE 2003; Audsley et al. 1997; Brentrup et al. 2004b; Peruzzi and Sartori 1997)	Modelled with CERES-EGC	Elaborated according to Audsley et al. (1997)	databases integrated in SIMAPRO (SimaPro 7.3 2012)

Crop management differences among the cropping systems are described in Table 1. It was assumed that farm transport of fertilisers and pesticides involved a 80.9 kW tractor either with a fertiliser spreader or pesticide

sprayer. All seeds for sowing were transported to the field within the seed drill with a 95.6 kW tractor. The same machinery is subsequently utilised to sow the various crops present in the rotation.
Harrowing in the PHEP system and mulching in the 50%GHG system was assumed to be carried out with a 80.9 kW tractor, together with pesticide treatments; while ploughing in PHEP was done with a 118 kW tractor. Cultivator pass in PHEP and roller pass in both systems were performed with the 95.6 kW tractor. Finally, a 180 kW combined harvester was assumed to be used for grain harvest together with a 10 Mg maximum load trailer and a 80.9 kW tractor.

Regarding fertilizer production, all data needed to account for raw material extraction, fertiliser manufacture and transport from raw material extraction sites to local storehouse were accounted with SimaPro 7.3 (2012). Fate factors for P and K fertilizers into the various environmental compartments were taken from Audsley et al. (1997) (Table 2).

The crop management of the two cropping systems analysed made use of several different pesticides: herbicides, molluscicides, insecticides and fungicides. Due to the high heterogeneity of pesticides and herbicides used, a common procedure was utilised to estimate the emissions to the air, water, and soil compartments, on the basis of the amount and type of active ingredient, which was also used to evaluate upstream impacts (Audsley et al. 1997; Brentrup et al. 2004a) (Table 2); while pesticide transport impact was computed on a total weight basis (Goglio et al. 2012).

3. Results

3.1. Simulation of field emissions

The emissions of reactive N (as N_2O, NH_3, NO and NO_3) simulated by the ecosystem model over Ile de France are given in Table 3, along with the annual change of soil organic C (SOC) for the two cropping systems due to soil mineralization. As expected, the 50%GHG system achieved higher C sequestration rates than the reference system (PHEP), with a 6-fold relative difference due to higher returns of crop residues to the soil. Surprisingly, N_2O emissions were 20.6% higher for the 50%GHG system in comparison with the PHEP system. NO emissions followed a similar pattern, being controlled by the same soil processes as N_2O.

Table 3. Simulated emissions of reactive N and soil C change for the two cropping systems over the Ile de France region. Data correspond to the spatially-weighted average over the 18 (50%GHG system) to 20-year (PHPE) simulation periods, with the 5-95% percentiles given in brackets.

Cropping system	**Emissions of reactive N (kg N ha^{-1} yr^{-1})**				**Soil organic C change (kg C ha^{-1} yr^{-1})**
	N_2O (direct)	NO	NH_3	Nitrate	
PHPE	0.29 (0.05 – 0.94)	0.44 (0.34 – 0.51)	0.43 (-0.11 – 1.40)	14.6 (4.17 – 42.34)	-140 (-540 – 95)
50%GHG	0.35 (0.05 – 0.98)	0.50 (0.15 – 0.62)	0.34 (-0.09 – 1.19)	13.0 (3.15 – 33.5)	1000 (210 – 1460)
Relative difference (50%GHG – PHPE)/PHPE	+20.6%	-13.7%	-20.7%	-11%	+ 610%

Conversely, the 50%GHG system presented 10 to 20% lower ammonia and nitrate losses than PHEP on average, due to the systematic presence of catch crops over the winter, which limited the build-up of nitrate and ammonium near the soil surface. Over the entire region and the 680 spatial simulation units (representing different combinations of soil properties and climate data series), Nr fluxes per hectare had large variability around their spatial means (Table 2), as their percentiles values showed. Except for the C variation rates, the distribution of the 5% percentiles and 95% percentiles largely overlapped between the two systems, implying that their differences were not significant.

3.2. LCA results

The 50%GHG system was successful in reducing some of the impacts analysed compared to the reference PHEP system. It abated GWP by 184%, air human toxicity by more than 2% and water chronic ecotoxicity by more than 3% (Table 4). Its GWP was actually negative, with a regional average of -2 Mg of CO_2 eq ha^{-1} and -22.5 kg of CO_2 eq GJ^{-1} of grain energy thank to soil CO_2 uptake (Tables 3 & 4). The minimum change in GWP among treatments was 6.66% corresponding to the difference between GWP of the 95% percentile of the 50%GHG system and the GWP of the 5% percentile for the PHEP system. However the 50%GHG system had 16% higher water chronic ecotoxicity and 7% larger water human toxicity and water acute ecotoxicity than PHEP system with both functional units (Table 4). For the other impact categories, differences were limited (1-2%) (Table 4).

Table 4. LCA results for the two cropping systems (50%GHG and PHEP) across the Ile de France region. Data correspond to the mean value for each impact category with the impact results estimated from the 5% and 95% percentiles of reactive N species and CO_2 emissions in brackets

		50%GHG	**PHEP**		**50%GHG**	**PHEP**
Impact category	**Category unit**	**(ha^{-1})**	**(ha^{-1})**	**Category unit**	**(GJ^{-1})**	**(GJ^{-1})**
Cumulative energy demand	GJ eq[a]	28.1 (28.1-28.1)	28.5 (28.5-28.5)	MJ eq[b]	317 (317-317)	322 (322-322)
Global warming 100a	Mg CO_2 eq	-1.99 (-3.86-1.27)	2.38 (1.36-4.25)	kg CO_2 eq	-22.5 (-43.5-14.3)	26.9 (15.4-48.1)
Acidification	kg SO_2 eq	16.5 (15.9-16.6)	16.2 (16.0-16.3)	g SO_2 eq	185 (179-188)	183 (181-185)
Eutrophication	kg PO_4^{-3} eq	21.4 (16.8-30.5)	21.6 (17.0-33.9)	kg PO_4^{-3} eq	0.241 (0.190-0.344)	0.245 (0.192-0.384)
Air human toxicity	m^3	2.43E7 (2.42E7-2.44E7)	2.48E7 (2.47E7-2.49E7)	m^3	2.74E5 (2.73E5-2.75E5	2.81E5 (2.80E5-2.82E5)
Water human toxicity	m^3	1.21E5 (1.21E5-1.21E5)	1.12E5 (1.12E5-1.12E5)	m^3	1.37E3 (1.37E3-1.37E3)	1.27E3 (1.27E3-1.27E3)
Soil human toxicity	m^3	258 (258-258)	251 (251-251)	m^3	2.91 (2.91-2.91)	2.84 (2.84-2.84)
Water chronic ecotoxicity	m^3	5.27E6 (5.27E6-5.27E6)	4.42E6 (4.42E6-4.42E6)	m^3	5.94E4 (5.94E4-5.94E4)	5.00E4 (5.00E4-5.00E4)
Water acute ecotoxicity	m^3	2.03E5 (2.03E5-2.03E5)	1.89E5 (1.89E5-1.89E5)	m^3	2.29E3 (2.29E3-2.29E3)	2.14E3 (2.14E3-2.14E3)
Soil chronic ecotoxicity	m^3	1.20E6 (1.20E6-1.20E6)	1.23E6 (1.23E6-1.23E6)	m^3	1.35E4 (1.35E4-1.35E4)	1.39E4 (1.39E4-1.39E4)

[a] 1GJ eq corresponds to 1 GJ of energy from difference sources
[b] 1MJ eq corresponds to 1 MJ of energy from difference sources

In terms of variability only for cumulative energy demand, GWP, and toxicity impacts, the impact distributions could be considered different (Table 3). For the other impact categories impact differences were not significant despite having different means because their impact distribution overlaps. Table 4 shows that some impacts are not affected by variability in reactive N species and CO_2 emissions including cumulative energy demand and all the toxicity impacts except water human toxicity.

4. Discussion

4.1. Field emissions

Regional modelling was mostly aimed at generalizing results obtained at the local scale in the Grignon trial to a larger area; an administrative region featuring about 1 Mha of cropland on which the two cropping systems could be applied. A major limitation in this exercise was that the spatial and inter-annual variability of management practices were ignored since all cropping operations occurred at fixed dates from one experimental site and a few years, regardless of the climatic conditions of the year and the soil types. In practice cropping operations depend strongly on soil conditions, weather and field conditions so that this assumption is quite questionable. Thus, these simulations can only convey a sense of the spatial and temporal variability associated with the diversity of soil and climate combinations occurring within a large region (through the means and percentiles in the emission fluxes), and their impacts on the performance of cropping systems. The implementation of decision rules (e.g. a balance-sheet method for N inputs in spring) would also be an interesting avenue to improve on this point (Bergez et al. 2010).

Modelled yields for the field trial in Grignon were within 5% of the observed values during the 2009-2012 seasons, thanks to the calibration of parameters related to crop phenology and dry matter partitioning. Regional means with the model were slightly lower than the yields obtained in the Grignon trial (e.g. 15% lower for winter wheat), but this could be expected since the soil in Grignon is a deep loam with a relatively high yield potential. In a previous modelling exercise in the same region (Ile de France), Gabrielle et al. (2014) found that simulated yields compared well with yield records for the 2000-2010 time period for winter wheat, but noted a possible over-estimation bias for rapeseed. Some of the crops present in the cropping systems tested here are relatively uncommon (e.g. faba beans), and have seldom been attempted with crop models. Little information was available in the literature to parameterize the CERES-EGC model for these crops, and their simulation therefore carried a high degree of uncertainty.

Simulated emissions of Nr were relatively small compared to estimates based on emission factors such as the Tier 1 methodology from IPCC guidelines for N_2O (De Klein et al. 2006). Given the average N application rates (around 100 kg N ha^{-1} yr^{-1}), the latter would have yielded about 1 kg N_2O-N ha^{-1} yr^{-1}, compared to the 0.05–1.00 kg N_2O-N ha^{-1} yr^{-1} obtained here. Such trend was already noted and discussed by Gabrielle et al. (2014) observing that a top-down study based on atmospheric inversion showed that emission factors were lower in the context of France for N_2O than the Tier 1 default values. Regarding the other gaseous emissions (NH_3 and NO), emissions were also in the lower end of values reported for cropland. Conversely, the rate of C sequestration simulated for the 50%GHG system is in the higher range of literature values. For instance, a recent review on C sequestration rates cited a maximum figure of 1 Mg C ha^{-1} yr^{-1} for set-aside (as a cropland management option) in cool-moist climates such as Ile de France (Smith et al. 2008). On the other hand, the slight C loss rate predicted for the reference system (PHEP) is consistent with values recorded after 10 years in a nearby arable plot under minimum tillage (Loubet et al. 2011).

4.2. Differences in LCA results across systems

Compared to PHEP, the 50%GHG system reduced GWP by 184% across Ile de France, with both functional units (hectare and GJ of harvested biomass). The original 50% target assigned to 50%GHG was then largely achieved. Even considering the worst-case scenario for this system (the 95% percentile) and the best case for the PHEP system (the 5% percentile), the abatement still amounted to 6.66%. This difference in GWP among cropping systems was mostly due to soil C changes. The slight increase in N_2O emissions with the 50%GHG system was largely offset by the decrease in CO_2 emissions (Brady and Weil 2002; Six et al. 2004). Negative GWP values (as obtained with the 50%GHG system) were observed in previous work in relation to tillage, with an effect on SOC dynamics. For instance, Six et al. (2004) evaluated a negative GWP in humid conditions of up to 2.07 Mg of CO_2 eq ha^{-1}, which is in agreement with the findings of this study. However the large range of the 5% and 95% percentile GWP values for both systems confirmed the high variability highlighted by Six et al. (2004). The GWP range for the PHEP cropping system was comparable with figures obtained by Kim et al. (2009) for continuous maize in the US, who estimated reactive N species and carbon storage with the DAYCENT model and integrated its results in a LCA. They reported GWP ranges of 17.3-56.1 kg of CO_2 eq GJ^{-1} and 1.87-5.19 kg of CO_2 eq ha^{-1}, respectively. However, the latter applied to only one crop whereas the systems evaluated here included 4 to 6 different crops, and to a different agro-ecological zone.

Most of the agricultural LCAs found in the literature rely on IPCC emission factors and simple models to evaluate reactive N fluxes, and disregard soil C dynamics (Brentrup et al. 2004b; Charles et al. 2006; Goglio et al. 2012; Nemecek et al. 2011a; Nemecek et al. 2011b; Pelletier et al. 2008). Despite these differences with the methodology proposed here, some of these studies were comparable with the data elaborated here for the PHEP system. For instance, Brentrup et al. (2004b) reported GWP ranges of 8.2-26.5 kg of CO_2 eq GJ^{-1} and 0.29-4.10 Mg CO_2 eq ha^{-1} for wheat using different fertilizer rates. For the same crop in Switzerland, Charles et al. (2006) reported GWP values of 2.42 Mg of CO_2 eq ha^{-1} and 22.4 kg of CO_2 eq GJ^{-1}. The PHEP system had slightly lower GWP values than those reported by Nemecek et al. (2011b), for Swiss intensive and extensive cropping systems involving grain cereals (with a range of 2.89-5.03 Mg of CO_2 eq ha^{-1}). Notwithstanding, the GWP of the

PHEP system was more in line with the integrated and organic management of cropping systems with grain cereals (with a 2.15-5.03 Mg of CO_2 eq ha^{-1} range), estimated by Nemecek et al. (2011a).

There were less pronounced variations across the two cropping systems for the other impact categories, except for chronic and acute water ecotoxicity and human toxicity to water (with more than 7% difference). The two systems had similar energy consumption, both on ha basis and GJ basis. This was due to two compensating factors: the substitution of most mechanical weeding and tillage in the PHEP system with frequent pesticide treatments in the 50%GHG system for crop protection, and a limited difference in fertilizer application rates. The latter are known to largely affect energy consumption of agricultural products (Brentrup et al. 2004b; Charles et al. 2006; Goglio et al. 2012).

The energy demand for both systems was slightly greater than reported by Kim et al. (2009) for maize cultivation with and without tillage in a series of locations in the Corn belt (with ranges of 14.2-27.2 GJ ha^{-1} and 143-224 MJ GJ^{-1}, respectively). Other research evaluated only non-renewable energy demand for intensive and extensive (11-22 GJ ha^{-1}) (Nemecek et al. 2011b) or integrated and organic (10-22 GJ ha^{-1}, Nemecek et al. (2011a)) cropping systems with cereal grains in Swiss conditions and showed at least 21.7% less energy consumption on ha basis than the present assessment.

Acidification potentials presented here were consistent with numbers obtained for maize by Kim et al. (2009) on GJ basis (184-531 g of SO_2 eq GJ^{-1}), but lower than the range the latter reported on a ha basis (22.4-53.0 kg of SO_2 eq ha^{-1}). The same pattern occurred with the estimates of Charles et al. (2006) for wheat in Swiss conditions (with a 17.8 kg of SO_2 eq ha^{-1} value). Conversely, our data were somewhat larger than the range reported by Brentrup et al. (2004b) for wheat cultivation in the UK (3.1-15.9 kg of SO_2 eq ha^{-1}).

The eutrophication potentials calculated here were larger than Kim et al. (2009) (>14.5 kg PO_4^{-3} eq ha^{-1} and >156 g PO_4^{-3} eq GJ^{-1}) for maize cultivation. Using simple models to estimate reactive N species, Charles et al. (2006) reported lower values than eutrophication potentials elaborated here for the two cropping systems (over 3.47 kg PO_4^{-3} eq ha^{-1} and 0.0032 kg PO_4^{-3} eq GJ^{-1}, respectively). However both studies (Charles et al. 2006; Kim et al. 2009), evaluated either maize or wheat while the present cropping systems included also rapeseed, barley, faba beans, triticale and several cover crops. Regarding toxicity impacts, the PHEP system had lower water ecotoxicity (both chronic and acute) and water and soil human toxicity than 50%GHG. These differences might arise from the different types of pesticides used in both systems. Soil chronic ecotoxicity impacts estimated in this research were within range with those estimated with EDIP 97 method in integrated and organic cropping systems (3.7 e4-6.19 e6 m^3 ha^{-1}) (Nemecek et al. 2011a) and for intensive and extensive cropping systems including cereals in Switzerland (0.10-3.23 e6 m^3 ha^{-1}) (Nemecek et al. 2011b). These discrepancies might be due to different climatic conditions, but also to other crops present in the cropping systems evaluated by Nemecek et al. (2011a; 2011b).

5. Conclusion

Compared to the reference system representing current practice (PHEP), the 50%GHG cropping system achieved lower environmental impacts for some impact categories, including GWP (by 184%), on average, whether on a hectare or GJ basis. Using the latter functional unit, which may be deemed more relevant to the primary function of the cropping systems (which is providing biomass for food, feed or energy purposes), the relative differences between the two systems varied between -184% and 16% for the various LCA indicators. This demonstrates the benefits of the 50%GHG system and validates its *ex ante* design, although the large variability around the mean indicators tends to mitigate these benefits at the regional scale.

The present work emphasized the interest of evaluating the performance of agricultural crops at the cropping system level since it is the only relevant scale for most of environmental impacts because of the interactions between crops and the interplay with climate conditions. In terms of time scale, field emissions of Nr and especially of carbon dioxide involve dynamics that are hardly detectable on an annual level. Thus, long field trial evaluation is a key factor to have a better estimation of these dynamics and their interplay with climatic conditions. Indeed, long term trial data allow the evaluation of real crop management in relation to seasonal effects, avoiding a mean crop management independent from climatic seasonality.

The CERES-EGC model proved to be a useful tool to evaluate cropping systems on a time scale relevant to capture the differences gradually occurring between the two crop management options tested here. It gave the possibility of generalizing the results obtained in one location to the surrounding administrative region, taking into account interactions with soil and climate factors. Despite the absence of decision rules to account for management x environment interactions, and the lack of data to test model outputs at this scale, this exercise proved useful to explore routes to the mitigation of environmental impacts of arable crops via a systemic approach.

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Sensitivity analysis in life cycle assessment

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ABSTRACT

Life cycle assessments require many input parameters and many of these parameters are uncertain; therefore, a sensitivity analysis is an essential part of the final interpretation. The aim of this study is to compare seven sensitivity methods applied to three types of case studies. Two (hypothetical) case studies describing electricity production: one shows linear and another shows non-linear behavior. The third case study describes a large (existing) case study of seafood production containing high input uncertainties. The methods are compared based on their results, i.e. variance decomposition and ranking of the input parameters. Results show that Sobol' sensitivity indices perform the best for all three case studies. The Sobol' method can be a useful method in case of non-linear LCA models or LCA models that include outliers.

Keywords: matrix perturbation, method of elementary effect, key issue analysis, random balance design, Sobol' sensitivity index

1. Introduction

A life cycle assessment (LCA) calculates the environmental impact of a product from cradle to grave. LCAs require many input parameters and many of these parameters are uncertain. A sensitivity analysis, therefore, is an essential part of the final interpretation. This is mentioned in the ISO standards for LCA, but no guidance is given on how to do or how to select an appropriate sensitivity analysis. A sensitive parameter is a parameter of which a change considerably influences the result, or that contributes to the variance of the output. A sensitivity analysis can help identifying parameters that should be known accurately before drawing conclusions, or identifying non-sensitive parameters for which the variance can be fixed in the region of its variance in order to simplify a model, also known as 'factor-fixing' (Saltelli et al. 2008).

Sensitivity analysis in LCA can be performed using a one-at-a-time approach (OAT), meaning that a subset of the input parameters are changed one at a time to see how much influence it has on the results. Although this approach has many advantages, e.g. it is easy to perform and to understand, this type of sensitivity analysis is time-consuming for a large system and might not consistently take all parameters into account and, therefore, could overlook possible sensitive parameters. Sensitivity analyses that consistently analyze the sensitivity of each parameter in the model are usually performed with sampling based approaches, such as Monte Carlo simulation, with an added procedure for variance decomposition.

In general we can differentiate between three types of sensitivity analysis: local sensitivity analysis (e.g. OAT); screening (e.g. method of elementary effect) and variance based sensitivity analysis or global sensitivity analysis (e.g. regression analysis). An overview is given in Table 1. The methods differ in their input requirements (e.g. knowledge about probability distribution function and parameter of dispersion) and type of output: either a ranking and/or variance decomposition of the input parameters.

Table 1. Types of sensitivity methods discussed in this paper.

Type	Method
Local	Matrix perturbation (MP); one-at-a-time approaches (OAT)
Screening	Method of elementary effect (MEE)
Global	Standardized regression coefficients (SRC); key issue analysis (KIA); random balance design (RBD); Sobol' indices (SME and STE)

For most of the sensitivity methods mentioned in Table 1, it is not yet known under which conditions they optimally perform, or if they can outperform the standard practices in LCA (i.e. OAT, MP, SRC, KIA). The aim of this study is to compare seven sensitivity methods applied to three types of case studies: one showing linear, another showing non-linear behavior and a large linear case study, but with large input uncertainties. The methods are compared based on their performance i.e. variance decomposition and ranking of the input parameters

2. Methods

2.1. Case studies

In this study we applied the sensitivity methods to two case studies of the production of 1 MWh electricity (the original version of the case studies appeared in (Heijungs 2002; Heijungs and Suh 2002). Both case studies consisted of two processes: fuel production and electricity production. The first case study produces electricity thereby using fuel (figure 1). In the second case study, electricity is in turn necessary for fuel production, creating a loop (figure 2). The parameters of the second case study are known to be locally (very) non-linear (Heijungs 2002). For both case studies we assumed that each input parameter is normally distributed with a standard deviation equal to 10% of the mean. Furthermore we assumed that the parameters are uncorrelated and independent. For both case studies the parameters are numbered as follows: 1 (production of electricity); 2 (use of fuel for electricity production); 3 (use of electricity for fuel production, which is zero in the first case study); 4 (production of fuel); 5 (emissions of CO_2 during electricity production); and 6 (emissions of CO_2 during fuel production).

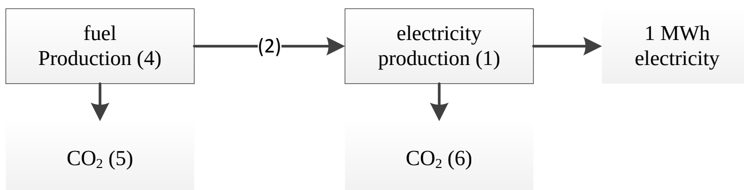

Figure 1. Case study 1: production of 1000 kWh electricity requires 200 liter fuel.

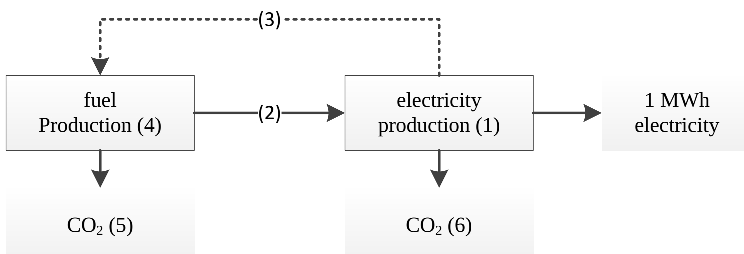

Figure 2. Case study 2: production of fuel requires electricity.

The third case study describes a trawler operating in the Northeast Atlantic, targeting mainly cod and haddock. The case study consists of 115 input parameters, describing e.g. production of vessel and gear, fuels, anti-fouling and cooling agents (figure 3). We assumed that the input parameters come with high uncertainty, each standard deviation of each parameter varies with 30% of the mean. All parameters are log-normally distributed.

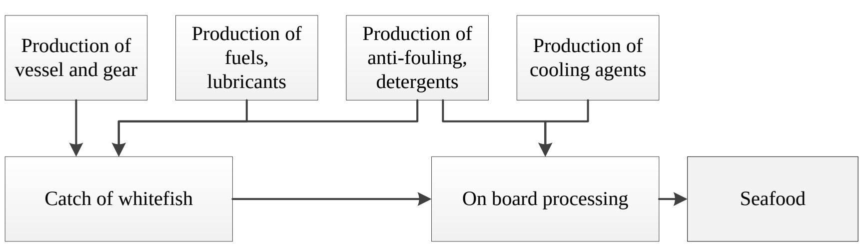

Figure 3. Case study 3: production of seafood.

The trawler takes trips of about ten to fourteen days, landing its catch in Tromso, Norway. The functional unit consists of 1 tonne landed whitefish.

2.2. Methods for sensitivity analysis

2.2.1. One-at-a-time

One-at-a-time (AOT) approaches take a subset of the input parameters, which are changed one at a time (either within its range or using an arbitrary value) to see how much influence it has on the result. The method is easy to perform and to understand, but this type of sensitivity analysis is time-consuming for a large system and, therefore, might not consistently take all parameters into account and could overlook possible sensitive parameters.

2.2.2. Matrix perturbation

Matrix perturbation (MP) as a method of (local) sensitivity analysis was introduced in LCA by Heijungs and Suh (2002). MP makes use of the first order partial derivatives as estimators of local sensitivity, which can be converted into relative multipliers. If the multiplier is larger, a change in the input parameter will change the result more than when the multiplier is almost zero. Information such as the type of distribution function or parameter of dispersion is not used. The result of this method shows how much the results will change if the input parameters are slightly changed (perturbed). This means that the multipliers predict the magnitude and direction of the change in the result of a small change around the original value of each parameter. A disadvantage of applying MP is that the method considers the model in its current configuration, and therefore the result (in general) only holds for small changes around the original parameter values. Moreover, information on which input parameter is quite certain and which is not (i.e. of the ranges of the parameters), is not used.

2.2.3. Method of elementary effects

The method of elementary effects (MEE) is a screening method that was originally designed by Morris (1991) and adjusted by Campolongo et al. (2007). MEE has been applied in LCA by Mutel et al. (2013) and de Koning et al. (2010). In order to apply MEE the ranges of the individual parameters are taken into account, where a range is defined as the upper and lower boundary of an input parameter. MEE can be seen as an extended one-at-a-time approach (Saltelli et al. 2008). MEE combines alternative values of each parameter (at predefined proportional steps in the range defined) and calculates the result. The difference between the original model and the new result of each combination is the elementary effect. Another indicator that can be calculated is the standard deviation of the elementary effect, which is an indicator for the interaction or non-linear effects: if the elementary effect of a certain parameter changes considerably for each run, the magnitude of the elementary effect depends on either the configuration of the model or the presence of nonlinear effects. MEE can be used as a precursor to the more computationally demanding sampling methods as regression. A disadvantage of this method is that the results are not an estimation of the actual variance decomposition.

2.2.4. Key issue analysis (Taylor expansion)

Key issue analysis (KIA) was introduced in LCA by Heijungs (2002) as a method for (analytically) determining the contribution to variance (or variance decomposition) by means of a first order Taylor expansion. KIA has been applied in LCA for example by Heijungs and Huijbregts (2004) and Heijungs et al. (2005). It combines the steepness of a function (described in section 2.2.1 above) with the variance of the individual parameters. KIA calculates the variance decomposition up to first order (or “main effects”), as covariance between input parameters are mostly unknown (Heijungs, 2002). To apply this method only the variances of the individual parameters are used. A disadvantage of KIA is that it does not produce a distribution function of the output, making it more difficult to compare two or more studies.

2.2.5. Standardized regression coefficients (using Monte Carlo sampling)

Standardized regression coefficients (SRC) are obtained from the slope of the line from least square fitting and estimate the contribution to output variance for each input parameter. Pseudo-random samples are taken from all input parameters and for each run the output is calculated. Subsequently, for each input parameter the regression coefficient is calculated; the coefficients are standardized with respect to their standard deviation. An advantage of calculating SRC is that it is commonly applied (in and outside) LCA, a disadvantage is that many runs are needed to calculate the variance decomposition.

2.2.6. Random balance design

The foundation of random balance designs (RBD) are from Cukier et al. (1978). In this study we use the format similar to Tarantola et al. (2006). RBD has been not yet been applied in LCA to our knowledge, although a very closely related method Fourier amplitude sensitivity test, has been applied by de Koning et al. (2010). Random balance designs estimate the contribution to variance by using Fourier transformations. A periodic sampling is applied and for each input parameter the Fourier spectrum is calculated, which is an estimate for the first order sensitivity index. A disadvantage of RBD is that only the main effect can be calculated.

2.2.7. Sobol' sensitivity index

The method by Sobol' (2001) assigns a sensitivity measure to each input parameter by calculating how much of the output variance can be allocated to each input parameter. The idea of the method is that a model can be decomposed into terms of increasing order, where the first order terms, also called the Sobol' main effects (SME), are equal to the contribution of variance caused by each input parameter to the output variance (Saltelli et al. 2010). The method also allows calculation of the interaction effects (variance caused by varying two or more parameters simultaneously) and the total effect index. The Sobol' total effect index (STE) gives the variance caused by the sum of the main and interaction effects of an input parameter. A disadvantage of Sobol's method is that many runs are needed to calculate the indices; hence the model is computationally expensive.

3. Results

3.1. Results of local methods

First the results of the local methods are presented, because it is not accurate to compare local with global methods, as they do not display similar information. Global sensitivity methods (or variance-based methods) include uncertainty information such as the variance into their results, while local methods estimate the change in the outcome based on the configuration of the (LCA) model at hand. The ranking of parameters by applying the local methods can be found in Table 2.

Table 2. Ranking of the parameters of the local methods for case study 1, 2 and 3, e.g. parameter 1 is most sensitive for case study 1 and 2. OAT: one-at-a-time approach; MP: matrix perturbation. The meaning of the parameters of case study 1 and 2 can be found in figure 2. FE: fuel use; EP: emission factor of fuel production; FP: fuel production; EC: emission factor of fuel combustion; LF: landed fish.

	Case study 1		**Case study 2**		**Case study 3**	
Rank	**OAT**	**MP**	**OAT**	**MP**	**OAT**	**MP**
1	1	1	2	1	LF	LF
2	5	5	1	2, 4	EC	EC
3	2	2, 4, 6	3	3	FP	FP
4	6		4	5	EP	EP
5	4		5	6	FE	FE
6			6			

For case study 1 both methods give similar results. For case study 2 the results differed slightly, although the actual values for parameter 1 to 4 were very close for both methods. Case study 3 gives also similar results.

3.2. Global methods: case study 1 electricity production

Applying the global methods to the first case study showed that the ranking of the input parameters were (almost) similar for each method (Table 3). The variance decomposition is given between brackets, but cannot be calculated for MEE as this method only gives a ranking of parameters. For example, in case of applying KIA parameter 1 is responsible for 56% of the output variance.

Table 3. Ranking of the parameters for case study 1, e.g. parameter 1 is most sensitive. MEE: method of elementary effect; KIA: key issue analysis; SRC: standardized regression coefficient; RBD: random balance design; SME: Sobol' main effect index; STE: Sobol' total effect index.

Rank	**MEE**	**KIA**	**SRC**	**RBD**	**SME**	**STE**
1	1	1 (56%)	1 (57%)	1 (58%)	1 (53%)	1 (58%)
2	5	5 (39%)	5 (37%)	5 (40%)	5 (33%)	5 (38%)
3	2	2, 4, 6 (1.6%)	2 (1.7%)	6 (2.5%)	4 (3.8%)	2 (1.7%)
4	6		6 (1.6%)	2 (2.4%)	2 (3.1%)	6 (1.6%)
5	4		4 (1.5%)	4 (2.1%)	6 (2.4%)	4 (1.6%)

KIA required only a single calculation, which means (in case of LCA) that these methods are computationally very fast. MEE required relatively few runs compared to the other sampling based methods. SRC, RBD and the Sobol' indices made use of sampling, SRC and the Sobol' indices requiring the largest number of runs. Although the individual contributions differed between methods, the overall picture is the same.

3.3. Global methods: case study 2 electricity production

Applying the methods to the second case study, we found that SRC, RBD and SME did not give reasonable results (Table 4), all parameters showed a contribution of about 0%. Although the ranking of the parameters is somewhat different, it should be noted that the difference in sensitivity between parameter 1, 2, 3 and 4 were very small in case of MEE and KIA (Table 4). Only the Sobol' total index (STE) explicitly indicated parameter 3 as begin responsible for most of the output variance. The variance decomposition given by KIA and STE are given in Table 4 (SRC, RBD and SME are not shown as they did not gave reasonable results). KIA shows the main effects, indicating that parameter 1-4 are responsible for 25% of the output variance, the STE shows that, taking all interaction into account between, e.g., parameter 3 and the other parameters in the model, that parameter 3 is responsible for almost 93% of the output variance.

Table 4. Ranking of the parameters for case study 2. MEE: method of elementary effect; KIA: key issue analysis; SRC: standardized regression coefficient; RBD: random balance design; SME: Sobol' main effect index; STE: Sobol' total effect index.

Rank	**MEE**	**KIA**	**SRC**	**RBD**	**SME**	**STE**
1	3	1 (25%)	-	-	-	3 (93%)
2	4	2, 3, 4 (25%)				2 (50%)
3	2	5 (0%)				4 (50%)
4	1	6 (0%)				1 (50%)
5	5					5 (0%)
6	6					6 (0%)

3.4. Global methods: case study 3 production of seafood

Applying the global methods to the third case study showed that the ranking of the input parameters were (almost) similar for each method (Table 5), just as for case study 1. The five most sensitive parameters are shown, that contribute more than 1% to the output variance. The variance decomposition is given between brackets.

Table 5. Ranking of the parameters for case study 3. MEE: method of elementary effect; KIA: key issue analysis; SRC: standardized regression coefficient; RBD: random balance design; SME: Sobol' main effect index; STE: Sobol' total effect index. FE: fuel use; EP: emission factor of fuel production; FP: fuel production; EC: emission factor of fuel combustion; LF: landed fish.

Rank	MEE	KIA	SRC	RBD	SME	STE
1	LF	LF (56%)	LF (46%)	LF (51%)	LF (55%)	LF (59%)
2	EC	EC (41%)	EC (38%)	EC (36%)	EC (38%)	EC (41%)
3	FP	FP (~1%)	FP (~1%)	FP (~1%)	FP (~1%)	FP (~1%)
4	EP	EP (~1%)	EP (~1%)	EP (~1%)	EP (~1%)	EP (~1%)
5	FE	FE (~1%)	FE (~1%)	FE (~1%)	FE (~1%)	FE (~1%)

An interesting difference with case study 1, now that we have increased the uncertainty of the input parameters, is that STE is able to explain more of the variance than SME. SRC underestimates the contribution of parameters LF compared to the other methods. The sum of SRC, RBD and SME equals approximately 90%, indicating that the LCA model contains outliers. The sum of the variance decomposition according to KIA however, still sums up to 100% and does not show the presence of outliers.

4. Discussion

We did not take correlations between input parameters into account. If correlations are not taken into account, the result of the variance decomposition might be over- or underestimated. Nevertheless, we think that the results in this paper will also hold for LCA models containing correlated input parameters.

5. Conclusion

The results of a sensitivity analysis in LCA are important because they can be used to identify parameters that can considerably change the result, and which might need further investigation. They can also be used to identify parameters that are responsible for most of the output uncertainty and therefore should be known accurately before presenting results. Based on this study, we prefer the use of matrix perturbation in case of assessing the local sensitivity as it is more consistent than OAT where only a subset of the input parameters can be analyzed. MEE is a useful method in case of large models as a precursor to more computationally expensive sampling based methods. In case of interest in variance decomposition, we recommend the use of the Sobol' sensitivity indices in combination with KIA. Sobol' indices allow the calculation of main, interaction and total effect indices and the total index can also be calculated when the case study at hand shows non-linear behaviour. When the LCA practitioner is only interested in the main effect in case of a linear case study, KIA can be used.

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LCA study of unconsumed food and the influence of consumer behavior

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ABSTRACT

In light of resource depletion and anthropogenic influences on the greenhouse effect, food waste has garnered increased public interest in recent years. The aim of this study is to analyze the environmental impacts of food waste and to determine to what extent consumers` behavior influences the environmental burden of food consumption in households (hereafter called 'use'). An LCA study of three food products is conducted and addresses the impact categories climate change (GWP100), eutrophication (EP), and acidification (AP). Primary energy demand (PED) is also calculated. For adequate representation of consumer behavior, scenarios based on various consumer types are generated. If consumer acts careless towards the environment, the use stage appears as the main hotspot in the LCA of food products. Moreover, results show that the avoidance of wasting unconsumed food can reduce the environmental impact significantly.

Keywords: food waste, consumer behavior, disposal options, environmental impact

1. Introduction

Food is an essential element of human life. The human body demands high-energy organic compounds to function. Furthermore, food contributes to mental, physiological and social comfort (Baccini and Bader 1996).

In the light of global resource depletion and anthropogenic influences on the greenhouse effect, and as these relate to shortages in developing countries, the environmental effects of food production and consumption as well as the influence of food waste in industrialized countries has gained public attention in recent years (Gustavsson et al. 2011; Thurn 2011). Throwing away edible food affects the environment all along the value chain via production, logistic and disposal processes (FAO 2012). This paper aims to examine the amount and effects of unconsumed food in German households, where "unconsumed food" is defined as food products which were still edible at the time of disposal.

Participant food waste diaries of the EU-project "GreenCook" (GreenCook 2011; Ludwig 2013), which were kept for a period of three months, and a review of scientific literature (Gruber 2013) was used to select three food products for further investigation. A life cycle assessment (LCA) was performed for each food, concentrating on the use stage. Results indicate how consumers can influence the environmental profile of food in the use stage. Food-related activities, such as purchasing, storage, preparation and disposal, are also analyzed via LCA. Finally, the burden on the environment of the whole life cycle of wasted edible food is determined.

2. Methods

This study follows the ISO 14040/44 life cycle assessment (LCA) guidelines (ISO 14040 2006; ISO 14044 2009). The LCA software GaBi 6 and related databases were used for LCA modelling (PE International 1992-2013). Environmental impacts were calculated according to the CML 2001 method. Climate change (GWP100), eutrophication (EP) and acidification (AP) were analyzed. Primary energy demand (PED) was calculated additionally.

2.1. Selection of food

Three food products – potatoes, milk and rice – were selected according to the following criteria: (1) high level of disposal, (2) a clear designation in the household diary, (3) no convenience products or food with unclear composition and (4) staple food without seasonal limitation for consumption. Moreover, the use phase should contain at least one process relevant to the energy demand, e.g. refrigerated storage or required cooking. The selected food items and the amounts wasted per person per year are compiled in Table 1.

Table 1. Selected food and wasted edible amount per person during one year (Ludwig 2013)

Food	Waste amount [kg/person·year]
Potatoes	15.1
Milk	11.6
Rice	3.1

2.2. System boundaries and functional unit

The entire life cycle, from cradle to grave, of potatoes, milk and rice was modelled. Germany is used as the geographical reference, with the exception of the agricultural production and industrial processing of rice: here China was chosen as reference country. The time frame is based on current production conditions and the data used reflects the state of the art. The functional unit (FU) was chosen to be 1 kg food disposed after the use phase.

The system boundary, including all relevant processes such as primary production, industrial processing, wholesale and retail, is displayed in Figure 1 for the three different foods. The following differences occur in the use stage. (1) Potatoes and rice are stored at ambient temperature. (2) In contrast to potatoes and rice, milk requires no energy mix prior to consumption in the use stage. Pasteurized whole milk was chosen and was assumed to be consumed uncooked. Pasteurization is done in the industrial processing stage.

The model also includes the consumption of food via digestion in the human body. Although the consumption of food is not part of the functional unit itself, it was considered as a reference for the environmental impact of different disposal routes. A direct comparison of the environmental impact of consumption and the different disposal options does not take place, because abstention from food consumption is no appropriate solution to reduce the environmental impact. The function of food is to supply the human body with energy rich organic compounds (Baccini and Bader 1996).

The disposal routes for the three foods were chosen according to the data from the household diaries (Ludwig 2013). Milk is mainly disposed of in the sink and is delivered to the wastewater treatment facility of a municipal sewage plant, while potatoes and rice are either disposed of with residual solid waste in waste incineration plants or with biological waste in composting plants. Composting was chosen for geographic reasons instead of fermentation in biogas plants, as it is a more common practice for the disposal of biological waste in Germany (Kern et. al 2010).

Capital goods were included in the background datasets for primary production, waste incineration, wastewater treatment and transportation processes. It was estimated that capital goods have a relatively low environmental impact, due to high mass flows during the life span of the infrastructure. Machines, buildings and infrastructure were excluded from use stage processes and from the composting plant model due to lack of complete and clearly assignable data. Also not taken into consideration were transportation of auxiliary materials, refrigerant emissions from cooling appliances in wholesale and retail as well as dishwashing in households. Food waste besides the use phase was not taken into account due to missing data on the wasted amount of specific food products during agricultural production, industrial processing, wholesale and retail (Kranert et al. 2012).

Allocation problems occur in processes that capture the utilization of various products at the same time. This applies to processes such as the storage of food in wholesale and retail. Mass allocation was conducted by computing the share in quantity of a product. The energy demand for the heating of the warehouse and supermarket and for the refrigerated storage of milk was allocated by mass to each respective food. Transport processes were also allocated by mass.

2.3 Data sources and data quality

Datasets from the GaBi database were used for designing and adjusting the models as far as possible (PE International 1992-2013). Missing data were complemented by scientific literature, with the exception of composting of biological waste, where primary data were collected and a model of a composting plant was built. All data used in the model were in the timeframe 2000 to 2013.

An LCA model developed by Muñoz et al. (2007) describes the biochemical transformation of food in the human body and was used in this study to model digestion processes. For the treatment of wastewater datasets from GaBi database were used (PE International 1992-2013).

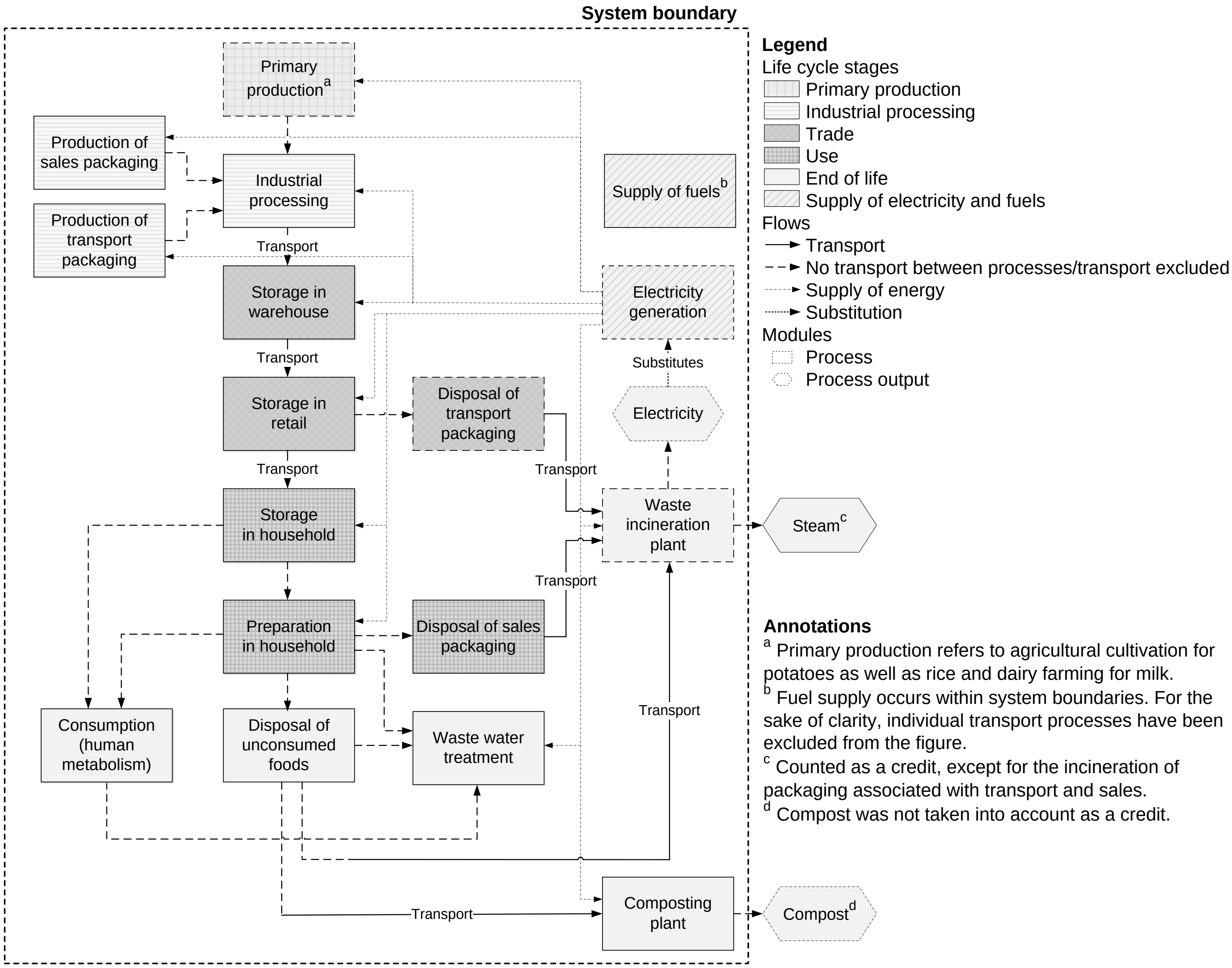

Figure 1. Schematic flow chart of the product system

2.4. Scenario definition

Scenarios based on three different consumer types were generated for adequate representation of consumer behavior during the use phase. (1) The base scenario (base-s) represents the “average consumer”. (2) The “environmental conscious consumer” (conscious-s) acts with ecological awareness due to practicing a lifestyle that minimizes damage to the environment by consciously keeping resource consumption and energy as low as possible. (3) The “careless consumer” (careless-s) remains indifferent to potential environmental impacts of his actions. According to these assumptions, parameters for the purchase, storage, and preparation of the food products were defined. The end-of-life disposal routes for the unconsumed food were set according to the household diaries. In base-s, it was assumed that food is not wasted but consumed. In conscious-s, the food waste was assumed to be separated from the general waste. Unconsumed food is disposed of in the organic waste bin. In careless-s, wasted food is disposed of with the residual solid waste.

2.5. Purchasing

Purchasing is done in the base-s and careless-s cases by car. The distance travelled from household to retail (and return) was set to 10 km according to Tengelmann (2009). In the conscious-s case, where purchasing is not done by a fuel-powered vehicle, the parameter is set to 0 km, because it is assumed that travelling by foot or bi-cycle has no impact on the environment. Shopping trips combined with other activities, e.g. travelling to work and shopping, were not taken into account as combined trips often lead to longer transport distances. Allocation among the different causes was not possible due to insufficient data.

In base-s and conscious-s, it was assumed that the consumer purchases several products, weighing a total of 10 kg, including the relevant food product. In careless-s it was assumed that consumer is badly organized and only one sales unit of the relevant food is bought. Here it was assumed that base-s and conscious-s plan and organize their shopping trips and purchase can also contain non-food products. Table 2 shows the purchase quantity for the scenarios and the weights of sales units including packaging.

Table 2. Weights of sales units of potatoes, milk and rice

Purchase quantity	**Base-s**	**Conscious-s**	**Careless-s**
Total purchase quantity [kg]			
Scenarios potatoes	10	10	2.51[a]
Scenarios milk	10	10	1.06[b]
Scenarios rice	10	10	1.01[c]
Share of considered food product [%]			
Potatoes	20.51	20.51	100
Milk	10.07	10.07	100
Rice	10.00	10.00	100

[a] 2.50 kg potatoes, weight of sales packaging 0.014 kg ; [b]1.03 kg milk, weight of sales packaging 0.03 kg; [c] 1.00 kg rice, weight of sales packaging 0.005 kg

2.6. Storage

Potatoes and rice were stored in households at ambient temperature; their storage contributed no environmental burden. The electric energy demand for storage of milk in the refrigerator [MJ/FU] was calculated by a parameterized equation (Eq. 1) based on (Nielsen et al. 2013; Sonesson and Janestad 2003; Geppert 2011). Parameters of Eq. 1 and the data used in the scenarios are listed in Table 3.

Caused by the interaction between consumer and kitchen appliances the electric energy consumption of the refrigerator is increased by about 60 % compared to the operation of a refrigerator under ideal conditions (Defra 2008). Energy losses may result from (1) door of the refrigerator left open too long, (2) irregular defrosting, (3) a dusty heat exchanger, (4) too warm location of the refrigerator, and (5) storage of hot dishes (Utopia AG 2013). Therefore the equation was complemented with the parameter e_K describing these energy losses.

$$E_{KS} = e_K \cdot \left(\left(\frac{E_{Kspez.}}{V_K} \cdot \frac{100}{n} \cdot V_P \cdot t \right) + \left(m_P \cdot c_V \cdot \left(T_A - T_K \right) \right) \right) \qquad \text{Eq. 1}$$

Table 3. Parameter description of Eq. 1 and parameter values used in the scenarios

			Value for scenario		
Parameter	**Description**	**Unit**	**Base-s**	**Conscious-s**	**Careless-s**
e_K	Parameter for interaction between consumer and refrigerator[a]	[-]	1.2	1.0	1.4
$E_{Kspez.}$	Specific electric energy demand of the refrigerator[b]	[MJ/d]	1.5	1.0	2.0
V_K	Capacity[b]	[dm³]	117	117	117
n	Rate of utilization of the capacity[a]	[%]	50	50	50
V_P	Volume of product for storage[c]	[dm³]	1	1	1
t	Duration of storage[d]	[d]	5	2	1
m_P	Mass of product for storage	[kg]	1.07	1.07	1.07
c_V	Specific heat capacity of the product for storage[e]	[MJ/kg·K]	0.00377	0.00377	0.00377
T_A	Temperature of the product at the beginning of storage[f]	[K]	292	292	292
T_K	Average temperature in refrigerator[g]	[K]	278	278	278
E_{KS}	Electric energy demand for the storage of milk	[MJ/FU]	0.56	0.16	1.61

[a] according to (Defra 2008); [b] based on (Siemens-Electrogeräte GmbH 2013); [c] calculated volume of one sales unit of milk, density of milk 1.03 kg/dm³ (Töpel 2004); [d] according to (FrieslandCampina 2013); [e] according to (Töpel 2004); [f] assumed to be ambient temperature; [g] according to (Utopia AG 2013)

2.7. Preparation

Potatoes and rice have to be cooked before consumption. Electric energy demand for cooking was calculated in the model based on (Oberascher et al. 2011) who analyzed cooking of potatoes in two different ways. (1) In the ideal case the potatoes were cooked in a pot with closed lid on the highest power setting of the stove up to the boiling point. Then, the heat was reduced. (2) In the non-ideal case the pot is not covered with a lid. The highest heat setting is chosen and not reduced after reaching the boiling point. The ideal case was taken as a basis for conscious-s, the careless-s is modelled after the non-ideal case and a calculated average of the two cases was set as base-s. The electric energy demand for cooking of potatoes and rice was considered to be equal (Table 4).

Table 4. Demand of electric energy for cooking of 1 kg potatoes or rice (Oberascher et al. 2011)

Scenario	Electric energy demand [MJ]
Base scenario (base-s)	1.4
Environmental conscious scenario (conscious-s)	0.7
Careless scenario (careless-s)	2.1

2.8. Disposal routes

After storage and if it is necessary preparation, the food is disposed. The disposal routes were set according to the household diaries and listed in Table 5 with the different scenarios. The model of the composting plant was built based on primary data from the German composting plant Kirchheim unter Teck (Kompostwerk Kirchheim unter Teck 2013).

Table 5. Disposal routes for the scenarios of potatoes, milk and rice (Ludwig 2013)

Scenario	Potatoes	Milk	Rice
Base scenario (base-s)	Consumption	Consumption	Consumption
Environmental conscious scenario (conscious-s)	Composting	Consumption	Composting
Careless scenario (careless-s)	Waste incineration	Sink/Waste water treatment	Waste incineration

3. Results

In Table 6 the results per functional unit (FU) of the LCA of potatoes are listed. The results of the use stage are displayed as a total and in detail for purchasing, preparation and the disposal of the sales packing via waste incineration. Results of the life cycle stages primary production and retail trade were similar for all scenarios, since the same assumptions were made in the model. Primary production and industrial processing (datasets out of the GaBi database by PE International 1992-2013) were summarized in the following to simplify matters and hereinafter called production. Retail trade was modelled in a similar way for potatoes, milk and rice. There were no differences assumed for primary production and retail trade for the different scenarios, thus the results were identical.

Table 6. Results per functional unit (FU) for LCA of potatoes

Life cycle impact assessment categories	Production[a]	Retail trade	Purchasing	Preparation	Disposal of sales packaging	Use (total)	End of life
GWP [kg CO_2-eq/FU]							
Base-s	0.059	0.029	0.174	0.206	0.014	0.394	0.473[b]
Conscious-s	0.059	0.029	0.000	0.094	0.014	0.109	0.480[c]
Careless-s	0.059	0.029	0.692	0.315	0.014	1.021	0.555[d]
EP [g PO_4^{3-}-eq/FU]							
Base-s	0.021	0.010	0.074	0.041	0.000	0.116	0.197[b]
Conscious-s	0.021	0.010	0.000	0.020	0.000	0.020	0.051[c]
Careless-s	0.021	0.010	0.295	0.062	0.000	0.357	0.045[d]
AP [g SO_2-eq/FU]							
Base-s	0.126	0.059	0.374	0.347	0.004	0.725	0.227[b]
Conscious-s	0.126	0.059	0.000	0.159	0.004	0.162	0.261[c]
Careless-s	0.126	0.059	1.489	0.531	0.004	2.023	0.061[d]
PED [MJ/FU]							
Base-s	1.330	0.406	2.587	3.623	0.009	6.220	1.035[b]
Conscious-s	1.330	0.406	0.000	1.649	0.009	1.658	1.253[c]
Careless-s	1.330	0.406	10.292	5.548	0.009	15.850	-4.770[d]

[a] includes primary production and industrial processing; [b] consumption; [c] composting; [d] waste incineration

The use and end-of-life stages of base-s dominate all considered impact categories and have the highest PED. In the conscious-s case, the main contributor is the end of life; in case of careless-s the use stage has the highest environmental impact. The negative value for PED (-4.770 MJ/FU) of careless-s is due to a credit of -5.599 MJ/FU for the generation of electric energy and steam from the incineration of waste. In the use stage, the purchase of food contributes the highest environmental impact in the base scenario as well as in careless-s. During fuel combustion, CO_2 and volatile organic compounds (VOCs) were emitted. These emissions result in a GWP for purchasing of 0.692 kg CO_2-eq/FU in careless-s and 0.174 kg CO_2-eq/FU in base-s. No environmental impact was associated with purchasing of conscious-s. The environmental impact of end of life scenarios were not compared to base-s, because of the function of food (see section 2.2).

Table 7 shows the results per FU of the LCA of milk in this study. The use stage is presented in detail for purchasing, storage and the disposal of the sales packaging via waste incineration. In contrast to the results of potatoes, the production is the main contributor to all impact categories and has the highest PED (32.8 MJ/FU) in all scenarios. In careless-s the use stage dominates the environmental burden. The purchase of milk has the highest environmental impact in the use stage of careless-s. The environmental impact of the storage of the milk is lower than of the purchase. With a GWP of 0.031 kg CO_2-eq/FU the disposal of the sales packaging is similar to the storage of milk in base-s with 0.032 kg CO_2-eq/FU. In conscious-s the GWP of the storage (0.009 kg CO2-eq/FU) is even 29 % lower. The contribution to the EP with 0.203 g PO_4^{3-}-eq/FU is for the consumption significantly higher than for the disposal of milk with 0.008 g PO_4^{3-}-eq/FU. This is caused by a greater amount of organically loaded wastewater resulting from consumption.

Table 7. Results per functional unit (FU) for LCA of milk

Life cycle impact assessment categories	Production[a]	Retail trade	Purchasing	Storage	Disposal of sales packaging	Use (total)	End of life
GWP [kg CO_2-eq/FU]							
Base-s	0.842	0.026	0.178	0.032	0.031	0.241	0.360[b]
Conscious-s	0.842	0.026	0.000	0.009	0.031	0.041	0.360[b]
Careless-s	0.842	0.026	1.665	0.091	0.031	1.788	0.006[c]
EP [g PO_4^{3-}-eq/FU]							
Base-s	0.357	0.009	0.076	0.006	0.003	0.085	0.203[b]
Conscious-s	0.357	0.009	0.000	0.002	0.003	0.005	0.203[b]
Careless-s	0.357	0.009	0.709	0.017	0.003	0.730	0.008[c]
AP [g SO_2-eq/FU]							
Base-s	1.361	0.043	0.383	0.053	0.021	0.457	0.235[b]
Conscious-s	1.361	0.043	0.000	0.016	0.021	0.036	0.235[b]
Careless-s	1.361	0.043	3.583	0.154	0.021	3.757	0.009[c]
PED [MJ/FU]							
Base-s	32.808	0.230	2.650	0.560	0.107	3.317	1.071[b]
Conscious-s	32.808	0.230	0.000	0.164	0.107	0.270	1.071[b]
Careless-s	32.808	0.230	24.769	1.613	0.107	26.489	0.040[c]

[a] includes primary production and industrial processing; [b] consumption; [c] waste water treatment

In Table 8 the results of the LCA per FU of rice are listed. The results of the use stage are displayed in detail for purchasing, preparation and disposal of the sales packaging via waste incineration. Production of base-s has an EP of 0.278 g PO_4^{3-}-eq/FU and AP of 2.623 g SO_2-eq/FU. In careless-s the use phase is the main contributor to all impact categories. Similar to the potato PED results, rice has a negative value for end of life, due to a credit of -5.599 MJ/FU for the generation of electricity and steam from the incineration of waste. In the use stage of careless-s, purchasing of rice dominates all impact categories and the PED with 6.398 MJ/FU, respectively. A closer look on the end of life shows that GWP of the disposal option composting (conscious-s) and of consumption (base-s) is similar with 0.480 kg CO_2-eq/FU and 0.602 kg CO_2-eq/FU, respectively, since both processes describe the biological degradation of organic compounds.

Table 8. Results per functional unit (FU) for LCA of rice

Life cycle impact assessment categories	Production[a]	Retail trade	Purchasing	Preparation	Disposal of sales packaging	Use (total)	End of life
GWP [kg CO_2-eq/FU]							
Base-s	0.152	0.089	0.182	0.215	0.043	0.440	0.602[b]
Conscious-s	0.152	0.089	0.000	0.103	0.043	0.146	0.480[c]
Careless-s	0.152	0.089	1.730	0.324	0.043	2.097	0.555[d]
EP [g PO_4^{3-}-eq/FU]							
Base-s	0.278	0.022	0.077	0.055	0.005	0.137	0.185[b]
Conscious-s	0.278	0.022	0.000	0.034	0.005	0.039	0.051[c]
Careless-s	0.278	0.022	0.737	0.076	0.005	0.817	0.045[d]
AP [g SO_2-eq/FU]							
Base-s	2.623	0.162	0.391	0.359	0.024	0.773	0.214[b]
Conscious-s	2.623	0.162	0.000	0.170	0.024	0.194	0.261[c]
Careless-s	2.623	0.162	3.722	0.542	0.024	4.288	0.061[d]
PED [MJ/FU]							
Base-s	2.346	1.470	2.702	3.646	0.041	6.389	0.976[b]
Conscious-s	2.346	1.470	0.000	1.671	0.041	1.713	1.253[c]
Careless-s	2.346	1.470	25.731	5.571	0.041	31.343	-4.770[d]

[a] includes primary production and industrial processing; [b] consumption; [c] composting; [d] waste incineration

Table 9 shows an extrapolation of the LCA results for the annual consumption per capita in Germany. LCA results of base-s were takenfor production, retail trade and the use stage.. For unconsumed potatoes and rice, a combination of the disposal options composting and waste incineration was created. Milk is only disposed via sink. The amount of unconsumed food is taken from Table 1. In the case of the consumed food results of base-s were taken for the end of life. According to the German Statistisches Bundesamt (2012), 56.6 kg potatoes, 101.3 kg fresh milk products and 5.4 kg rice were consumed per capita in 2012. No data was available for consumed milk, therefore the amount of fresh milk products was used.

Through avoidance of waste from unconsumed potatoes, a general consumer can reduce GWP by approximately 22 %, EP by 13 %, AP by 20 % and PED by 16 %, respectively. In the case of milk, GWP, EP, AP and PED can be reduced by approximately 10 %. Avoiding unconsumed rice waste reduces emissions and PED by 30-37 %. Table 9 illustrates the avoided environmental impacts, as compared with the environmental impacts of a diesel-powered car. Considering a family of five, the avoided GWP of unconsumed potatoes corresponds to a route halfway across Germany (approximately 500 km). The GWP resulting from unconsumed milk is equivalent to about 440 km and the GWP of unconsumed rice corresponds to about 120 km.

Table 9. Extrapolation of LCA results to the annual consumption per capita in Germany (Statistisches Bundesamt 2012; Ludwig 2013)

Food product	GWP [kg CO_2-eq/FU]	EP [g PO_4^{3-}-eq/FU]	AP [g SO_2-eq/FU]	PED [MJ/FU]
Potatoes				
Unconsumed (wasted edible food)[a]	15.12	2.95	16.21	93.78
Consumed	54.02	19.45	64.39	508.89
Milk				
Unconsumed (wasted edible food)[a]	12.89	5.31	21.62	420.86
Consumed[b]	148.81	66.25	212.32	3791.25
Rice				
Unconsumed (wasted edible food)[a]	3.56	1.50	11.70	35.10
Consumed	6.92	3.36	20.37	60.38

[a] savings potential of avoiding unconsumed food; [b] annual consumption of fresh milk products per capita in Germany

4. Discussion

The results of this LCA study are hardly comparable to previous studies. Differences from previous studies include different system boundaries and levels of detail for examination of the life cycle stages as well as the representation of consumer behavior in the model. The consumer modelling included four parameters: (1) distance travelled for purchasing, (2) purchased amount of food products, (3) storage and preparation of food in households and (4) amount of unconsumed food in households. An evaluation of 36 LCA studies of food products, which can be found in Gruber (2013), showed that no general conclusions can be drawn as to which life cycle stage of different food products is most important as the level and type of environmental impact from individual life cycle stages varies according to the considered food product. In the current study, however, the most important factor influencing environmental impact is consumer behavior as identified by the authors. This is clearly seen when looking at the results of the different scenarios of the highly-processed staple products milk and rice. Depending on how the consumer behaves during the use stage, the primary production including the industrial processing or the use stage dominates the environmental impacts and the PED.

Modelling assumptions and estimations were based on literature data, which must be critically examined. Therefore, this study has some limitations. First of all, data gaps on the food losses of individual food products during the whole life cycle exist, as this is a seldom examined issue in LCA and the collection of this data requires much effort. The end-of-life inventory data for the waste incineration plant and the municipal wastewater treatment plant could are not adjusted for the input of a specific food product. In the case of waste incineration, the process describes an incineration of mixed municipal solid waste. The input flow of the waste water treatment plant is polluted municipal sewage water. Neither process reflects the specific composition of the emissions related to the disposal of the individual food products. The LCA model of the composting plant describes the composting of mixed organic waste and therefore also does not reflect the specific emissions of the disposal of individual food products. Due to this insufficient data availability, the comparability of the disposal options in this study is limited.

Despite the limitations of the statement of this study regarding the end of life, there is no doubt that the environmental impact can be significantly reduced if the amount of wasted unconsumed food decreases. Additionally, the environmental impact can be significantly decreased in the use stage by an environmentally conscious consumer. Purchasing has the highest effect on the evaluated impact categories and on the PED. The reason for this is the fuel consumption with its related release of emissions. The mass-based allocation of purchased food products transported by car must be examined. For example, the transport of 20 kg of products by car does not require significantly more fuel than the transport of 1 kg of products. Nevertheless, in light of the goal and scope of this study it makes sense to assign the environmental impact to consumers' behavior. The purchase of several items in one shopping trip saves additional trips and related fuel, and thereby reduces the environmental impact. Additionally, consumers can reduce the resulting emissions by decreasing the electric energy demand, particularly where food storage is concerned. For example, consumers can take the following measures: (1) select a refrigerator with a high energy efficiency class, (2) use a refrigerator of the appropriate size and use it often, (3) only open the refrigerator when necessary, (4) do not store warm dishes in the refrigerator, (5) defrost regularly, (6) situate the refrigerator in a cool location, (7) set the temperature to an optimal setting (about 7 °C) and (8) clean the heat exchanger on a regular basis. Furthermore, an energy-conscious handling of kitchen appliances while cooking reduces the overall environmental impact of the use stage. The consumer can decrease the electric energy demand for cooking by covering the pot with a lid and reducing the heat setting of the stove. Since each food processing step requires resources and leads to the release of emissions, consumers can significantly decrease the environmental impact of food by avoiding food waste in terms of unconsumed and also potentially energy-intensive prepared food.

When evaluating the environmental impacts from cradle to grave of food products, it has to be taken into account that food supplies the energy required by the human body and delivers many health benefits beyond energy and nutrition (Roy et al. 2009). Considering the results of this study, it may seem that stopping or minimizing food consumption would be the easiest way to reduce the associated environmental impacts. However, given that the function of food is to provide energy for the human body, eliminating consumption is not a reasonable possibility. Therefore, it is not appropriate to compare the metabolic-based emissions of the consumption with the disposal of food products, even though the human body has been found to be an important source of emissions in GWP and EP (Muñoz et al. 2007) and therefore should be included when identifying the life-cycle hotspots of a food product. In this study the consideration of consumption was necessary because the overall environmental impact of food consumption was assessed, taking into account both the consumed and unconsumed amount of each individual food product.

5. Conclusion

Results of this study show that measures for reducing the environmental impact of food consumption must take place at different levels. One of these levels is the life-cycle use stage, as here the consumer can decide the most efficient way to reduce associated emissions. Emissions from the use stage can be reduced by environmentally responsible consumer behavior. Together, the prevention of waste in terms of unconsumed food can significantly decrease the impact on the environment.

Due to a lack of understanding and awareness of environmental issues, the average consumer often misses opportunities for beneficial environmental behavior (Vázquez-Rowe et al. 2013). Through projects and educational campaigns such as "GreenCook" (GreenCook 2011) attempts are being made to encourage consumers to think and act proactively.

The influence of consumer behavior on the LCA results has been found to be important. The life-cycle use stage of food products should not be overlooked in LCA studies. It is important to include food waste in the entire environmental assessment and not only in the use phase. Further research must be conducted to represent consumer behavior more accurately in LCA. To enable comparison among results, the LCA community needs to develop a common method for modelling consumer behavior. Moreover, end-of-life data is required for modelling waste disposal emissions in more detail.

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Effect of dietary change on greenhouse gas emissions and land use demand – The state of knowledge in 2014

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ABSTRACT

In combination with technical advances in agriculture, human dietary change is suggested to be necessary to reduce the environmental impact of the food system. In this paper a systematic review, assessing the climate and land use impacts of dietary change, is performed. The aims are to evaluate the scientific basis of dietary scenario analysis and to estimate the potential of reducing greenhouse gas emissions and land use demand by changing the composition of the diet. The review includes 14 peer-reviewed journal articles and in total 55 scenarios assessing the greenhouse gas emissions and land use demand of different diets. The results suggest that dietary change, in areas with affluent diet, could play an important role in reaching environmental goals, with up to 50% potential to reduce greenhouse gas emissions and land use demand of the current diet.

Keywords: Review, diet, scenario, climate, land use

1. Introduction

Global food production, accounts for around 30% of total anthropogenic greenhouse gas emissions (GHGE), occupies more than a third of the world´s land surface (Garnett 2011), and is identified as a great threat to the environment (EC 2006). In combination with technical advances in agriculture, changes towards more sustainable eating patterns are suggested to be a necessary to meet environmental targets (Garnett 2011).

Knowledge of sustainable food consumption is increasing with the growing number of environmental and life cycle assessments of foods, meals, and complete diets. A method commonly used to assess the impact of different dietary patterns is dietary scenario analysis. The methodological approach of dietary scenario analysis can have decisive effect on the final results. To draw general conclusions on which dietary changes that can promote a development towards more sustainable food consumption, therefore, requires that results from several studies are analyzed and compared. However, so far, few syntheses of studies assessing the environmental impact of diet have been performed.

To synthesize the state of knowledge, this paper provides a systematic review of research articles which assessed the environmental impact of dietary scenarios. The objectives are to: i) evaluate the scientific basis of scenario analyses assessing the impact on GHGE and land use demand (LUD) of human dietary change, ii) estimate the potential of reducing GHGE and LUD via dietary change and iii) identify current gaps of knowledge. The paper can be used as an overview of the state of knowledge and evidence base of sustainable food consumption in the year of 2014.

2. Method

2.1. Literature search strategy

In order to ensure scientific quality and minimize the risk of bias, the study design and analysis of this review follows the PRISMA Statement protocol (Moher et al. 2009).

The literature search was performed in February 2014 with the use of Web of Knowledge (ISI), Scopus and Google Scholar. To assess the effect of human dietary change on GHGE and LUD, the terms: ‘diet’ or ‘food’ and ‘scenario’ were combined with the terms ‘climate’ or ‘greenhouse gas’ or ‘land’ or ‘sustain*’. In addition, related and relevant articles found in reference lists were reviewed. Articles included in this review meet the following six inclusion criteria: i) English-language publications; ii) published between 2005 and February 2014; iii) dietary scenario analysis is performed for a complete diet; iv) quantitative estimates of the effect on GHGE and/or LUD of human dietary change are provided; v) published in peer-reviewed scientific journals; vi) results are compared against reference scenarios of current (1990-2010) average food consumption in a specified population. Determination of articles that meet the inclusion criteria was made based on information available in titles and abstracts of the articles. In total, 14 articles that fulfilled the inclusion criteria were identified (Fig 1).

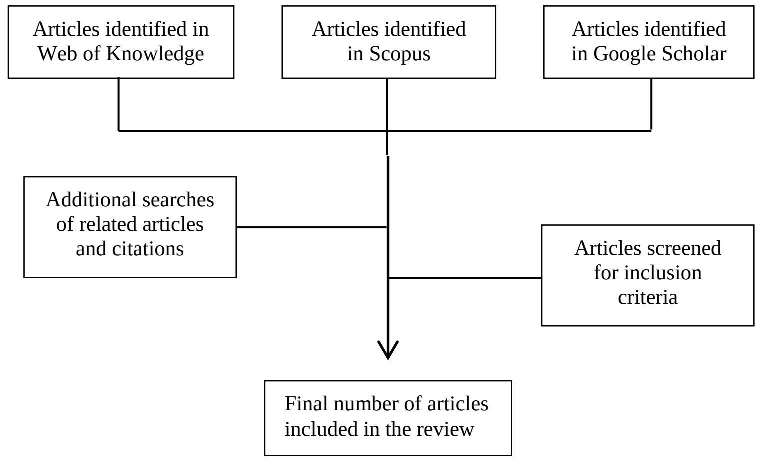

Figure 1. Literature search and selection of articles in the review

2.2. Synthesis of results

Depending on the dietary composition, scenarios were categorized into healthy diets, diets in which meat is partially replaced by plant-based foods/mixed foods/dairy products, diets in which all ruminant meat is replaced by pork and poultry, diet with balanced energy intake, vegetarian diets and vegan diets. The potential to reduce environmental impact is reported as the relative change in GHGE, expressed as tons of carbon dioxide equivalents (tCO_2e) per person per year, and LUD, expressed as square meter (m^2) per person per year, compared to the reference scenarios used in the respective studies.

3. Results

3.1. Located literature

In total, 14 articles that fulfil the inclusion criteria were identified in this assessment. Out of these 14 articles, two investigated both the effect on GHGE and LUD, two the impact of LUD only, and ten the impact of GHGE only. Five articles were published between 2009 and 2011, and nine articles between 2012 and Feb 2014 (Table 1).

In the majority of the studies the main focus is on the environmental impact of reducing or changing the consumption of meat and animal-based food (Table 2). In all articles except for Pathak et al. (2010) the effect of dietary change is studied in European populations characterized by having an affluent diet.

3.2. Potential to reduce GHGE

The impact of dietary change on GHGE from the diet is summarized in Table 2. Completely avoiding all animal-based products (vegan diet) or all meat (vegetarian diet) provides the largest potential for reducing GHGE from the diet, followed by scenarios of replacing ruminant meat by pork and poultry and eating a healthier diet.

3.3. Potential to reduce LUD

The impact of dietary change on LUD from the diet is summarized in Table 2. According to the results, a change to vegan or vegetarian diets has the largest potential to reduce the demand for agriculture land, followed by changing to a healthier diet and diets in which meat is partially replaced by plant-based food.

Table 1. Articles included in the review

Author	**Publication year**	**Environmental indicator**
Van Dooren et al.	2014	GWP, LU
Hoolohan et al.	2013	GWP
Saxe et al.	2013	GWP
Temme et al.	2013	LU
Aston et al.	2012	GWP
Berners-Lee et al.	2012	GWP
Macdiarmid et al.	2012	GWP
Meier and Christen	2012	GWP, LU
Vieux et al.	2012	GWP
Fazeni and Stenmüller	2011	GWP
Tukker et al.	2011	GWP
Arnoult et al.	2010	LU
Pathak et al.	2010	GWP
Risku-Norja et al.	2009	GWP

Table 2. Summary of results

Scenario	**Reduction of GHGE**		**Reduction of LUD**	
	(%)[a]	**(n)**	**(%)[b]**	**(n)**
Vegan diet	25-55	6	50-60	3
Vegetarian diet	20-55	9	30-50	2
Meat partially replaced by plant-based food	+5-0	2	15	1
Meat partially replaced by dairy products	0-5	2	-	0
Ruminant meat replaced by monogastric meat	20-35	2	-	0
Meat partially replaced by mixed food	5	2	-	0
Balancing energy intake and expenditure	0-10	2	-	0
Healthy diet	0-35	19	15-45	5

[a]Effect of dietary change on GHGE from the diet, in % of reduction in GHGE of current average diets. [b]Effect of dietary change on LUD, in % of reduction in total demand of agriculture land of the average diet. n = number of scenarios. "+" indicate an increase in GHGE alt. LUD.

4. Discussion

This review is, to our knowledge, one of the first to systematically assess the current state of knowledge of the environmental impact, expressed as changes in GHGE and LUD, of dietary change. The review includes peer-reviewed journal articles published over the past ten years.

4.1. Scientific basis

This review located 14 articles that met the inclusion criteria defined in this review. In accordance with what has been shown in Heller et al. (2013), this study illustrates that life cycle assessments (LCAs) of food is an expanding research field. Nine of the articles were published during just the two last years. Although there are still gaps in knowledge, the increased number of publications in this area has significantly contributed to a better understanding of sustainable production and consumption of food.

4.2. Potential to reduce GHGE

The results show that the potential to reduce GHGE from food consumption through dietary change can be substantial in regions with affluent diets. The reduction potential seems mainly to be dependent on the amount and type of meat and animal products included in the diet. Diets in which all meat and/or meat products are removed have the lowest GHGE. However, a healthier diet including meat can, according to the results, reduce the GHGE of the diet up to 35%. The impact is, however, largely dependent on what is considered to be a healthy diet, and in five of the 19 healthy dietary scenarios the reduction potential is less than 10%. The amount of red meat, and especially ruminant meat allowed in the healthy diets seem to be the decisive parameter for the climate impact of the diet. Another reason why the results vary is that some of the healthy dietary scenarios are based on organic production, which may lead to increased GHGE compared to conventional production system (Saxe et al. 2013). The difference in climate impact between different types of meat is also demonstrated by the results from the studies including scenarios where meat consumption is reduced or changed. Replacing all ruminant meat by poultry and pork can reduce the GHGE by up to 35%. Moderate reduction (up to 20%) in total meat intake (including white meat), in contrast, seems to have a negligible effect. In addition, the climate impact of the diet is, to a large extent, dependent on which foods that replace the meat, therefore, consumption of meat substitutes with high climate impact, such as cheese and air transported fruit and vegetables, should be restricted (Carlsson-Kanyama and Gonzalez 2009). Only eating necessary amounts of food has been identified as another priority measure to reduce GHGE from the diet (Garnett 2011) that also would be beneficial for health. Balancing the energy intake and expenditure can, according to the results in this review, reduce the climate impact of the diet by 0-10%, depending on the assumed energy requirements.

The GHGE from the reference scenarios, i.e. the current average diets in the studied populations, ranged from 0.9-1.7 and 1.5-3.2 tons (1.0 tons for Indian diet) of CO_2e per capita per year in the studies accounting for emissions up to farm gate and retail, respectively. The annual GHGE for the average EU citizen are around nine tons of CO_2e (EEA, 2012), which means that food consumption is responsible for about 15-35% of the total climate impact. Based on the these figures, the potential to reduce the total per capita GHGE through dietary change is about 3-20% for a transition to a vegan or vegetarian diet and up 12% by a transition to either a healthier diet with restricted intake of red and ruminant meat, or a diet in which the meat content partially been reduced and/or red or ruminant meat has been substituted by white or monogastric meat.

4.3. Potential to reduce LUD

Also the potential to reduce the LUD from the diet through dietary change may be considerable. It should, however, be kept in mind that the impact on LUD in this paper is based on only four articles. The potential to reduce the LUD of the diet appears to be largely dependent on the amount of ruminant meat consumed. Substituting all meat with plant-based food can, according to the results, reduce the land demand from the diet by up to 60%. According to Audsley et al. (2010) a replacement of 75% of the ruminant meat with pork and poultry can reduce the land demand by 40%. Replacing half of the consumption of pork and poultry with plant-based food would, on the contrary, only reduce the LUD by 5%. A healthy diet including meat may therefore also have a large potential to free land, if the consumption of red meat is limited. Diets including ruminant meat have previously been suggested to increase the number of people that can be fed from the same land area compared to vegan diets up to the point that land limited to pasture and perennial forages has been fully utilized (Peters et al. 2007). However, maximum output of food is necessarily not the primary objective, given that released land also can be used for bioenergy production, for example (Fazeni and Steinmüller 2011). Either way, as will be discussed further, differentiation between types of land is essential to fully understand the effect of diet on LUD.

The LUD of the reference dietary scenarios ranged from 1400-2100 m^2 per capita. This can be compared to the current global per capita availability of agriculture land which is about 7000 m^2 (divided between approximately 30% arable land and 70% pasture) if global croplands are assumed to be distributed equally across the population.

4.4. Identified research gaps

4.4.1. Differentiation on individual, regional and social level

The general approach to study the impact of dietary choices by using scenario analysis is to use a reference scenario based on the average per capita consumption in the population studied. Since consumption patterns and nutritional requirements differ depending on, for example, gender, age and physical activity level, it would be interesting to see more research on specific groups of the population. It is also noteworthy that all articles reviewed, except one, study the impact of dietary change in European countries/regions characterized by having affluent diets. To understand the impact of dietary change in a broader and global perspective similar studies are required in countries/regions with different habits, culture and conditions.

4.4.2. Differentiation of plant-based scenarios

Previous findings suggest that that plant-based food consumption based on self-selected diets tend to have a higher climate impact compared to plant-based consumption in hypothetical scenarios (Vieux et al. 2012). In plant-based hypothetical dietary scenarios, meat is often replaced by unprocessed foods such as pulses, cereals, salads, vegetables, fruit, nuts and seeds. Vegetarian diets are in general characterized by a higher proportion of these food groups (Craig 2010; Key et al. 2006), however, processed plant-based meat substitutes (e.g. processed soy, quorn, tofu, and tempeh) represent an increasingly important component of modern plant-based diets. The environmental impact of such processed vegetarian meat substitutes has so far only been investigated in a limited number of studies (Blonk et al. 2008; Davis et al. 2010; Finnagan 2010a; Finnigan et al. 2010b; Leuenberger et al. 2010; Nijdam et al. 2012; Nonhebel and Raats 2007; Xueqin and Ierland 2004). The results indicate that these products may have relatively high energy demands due to the higher degree of processing but a lower climate and overall environmental impact, in comparison to most types of meat. Few of the reviewed articles specify that these types of processed meat substitutes are included in the dietary scenarios. The potential and limitations for reducing the environmental impact of the diet through increased consumption of this group of food products requires further analysis.

4.4.3. Differentiation of agricultural land

Current global food supply is mainly dependent on cultivated land (Johansson 2005) why the pressure on agricultural land is especially intense on cropland. Previous studies indicate that dietary change, in particular, has the potential to free pasture land (Hallström et al. 2011). Of the land released through reductions and changes in meat consumption, for example, only 5-10% is estimated to consist of cropland (Hallström 2013). Others suggest that replacing beef with pork and poultry even may increase the total demand of cropland (Audsley et al. 2010). A net gain in cropland is also not obvious if consumption of dairy products is replaced by plant-based food or when monogastric meat is replaced by processed vegetarian meat substitutes (Audsley et al. 2010; Stehfest et al. 2009).

If the distinction is not made between different types of land, there is thus a risk of overestimating the land areas suitable for agriculture that can be released by reducing ruminant meat consumption as only a limited share of pasture land is suitable for cultivation. To avoid a situation where demand for agriculture land is exported to other countries where it might increase the risk for deforestation and other negative impacts connected to increased land use pressure, it may also be of interest to in a greater extent distinguish between domestic and foreign land use in dietary scenario analysis.

4.4.4. Accounting for uncertainty

Despite the knowledge of the uncertainty related to environmental and life cycle assessments, the environmental impact of dietary scenarios is in general reported in absolute numbers without standard deviations. This is questionable as it makes it difficult to evaluate the reliability of the results. According to the ISO standard, the interpretation phase in LCAs should include an evaluation of the completeness, sensitivity and compliance of the analysis (ISO 2006). This is required in order to help the reader to determine what conclusions can be drawn from the results and would be useful also in dietary scenario analysis.

4.5. Strengths and limitations

To minimize bias, this review includes only peer-reviewed journal articles selected by the use of predefined inclusion criteria. The aim has been to assess the articles with a high level of objectivity and transparency. A limitation with the study is that only a small number of articles which met the inclusion criteria were located. The limited number of articles can partly be explained by the novelty of the research field but is also due to the narrow inclusion criteria which excluded several relevant articles (Audsley et al. 2010; Eshel and Martin 2006; Kastner 2012; Macdiarmid et al. 2011; Marlow et al. 2009; Westhoek et al. 2011; Popp et al. 2010; Powell and Lenton 2012; Wirsenius et al. 2010; Stehfest et al. 2009). Relevant publications and data would perhaps also be found in non-English publications.

In this review, the majority of articles which quantify GHGE from the diet exclude emissions coming from direct and indirect land use change. The exceptions are Meier & Christensen (2012) and Holohan et al. (2013) who include GHGE from deforestation resulting from livestock supply chains. Greenhouse gas emissions from direct and indirect land use change are suggested to have substantial impact on the climate impact from agricultural products (Cederberg et al., 2011; Ponsioen and Blonk, 2012; Schmidinger & Stehfest, 2012), which means that the results on GHGE in this review may be underestimated.

The climate impact of diet is quantified based on the global warming potential (GWP) of GHE. In the fifth IPCC assessment report published in 2013 (Myhre et al. 2013) the GWP of methane over a time horizon of 100 years was increased from previously 25 to 34 kg CO_2e per kilograms of emissions. This review includes articles published before the new IPCC report was published and therefore use the lower GWP for methane in their calculations. This means that the climate impact of diets containing ruminant meat is likely to be higher than the results shown by this review.

In this paper the environmental impact of dietary scenarios is assessed only based on the emissions of GHGE and demand of agriculture land. These aspects can often, but not always, serve as indicators of other environmental impact categories such as eutrophication, acidification and loss of biodiversity (Rockström et al. 2009; Röös et al. 2013; van Dooren et al. 2014). However, for a full assessment of the environmental impact of the diet other environmental impact categories also have to be included. Within the wide concept of sustainable food production and consumption also several other aspects, of ecological, social and economic dimensions are included (FAO 2013). These aspects, however, go beyond the scope of this paper. In future studies interdisciplinary and holistic assessments of the diet which include more sustainability aspects are required.

5. Conclusion

This systematic review evaluates the potential of dietary change as a measure for more sustainable food systems. The results suggest that dietary change, in areas with affluent diet, can play an important role in reaching environmental goals, with up to 50% potential to reduce GHGE and LUD of the current diet. The reduction potential mainly depends on the amount and type of meat included in the diet but also on the environmental performance of the food substituting meat. In future research interdisciplinary and holistic assessments of the diet including more sustainability aspects are required. Improved knowledge is also needed on the uncertainty in dietary scenario studies, the environmental impact of substitutes and complements to meat, and the effect of dietary change in different groups of populations and geographical regions.

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Critical review of allocation rules – the case of Finnish rainbow trout

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ABSTRACT

In life cycle assessments (LCAs) one of the fundamental methodological choices that needs to be made is allocation, especially in multiple-output systems. The difficulty with allocation is that it usually has an ambiguous basis and, thus, damages the credibility of LCA results. Allocation solutions are easily influenced by perspectives and worldviews of the analyst, and there are arguments as to whether, for instance, allocation methods, such as economic, biophysical or mass-based allocations, are suited to different case studies and decision-making situations. Therefore, rules for different multiple-output situations are needed. While there are LCA guidelines that provide rules for different multiple-output situations, they do not fulfill their purpose, being susceptible to interpretation and/or enabling different allocation choices. This is demonstrated here with an attributional LCA case study of one tonne of Finnish rainbow trout fillet.

Keywords: LCA guidelines, allocation, harmonization, rainbow trout

1. Introduction

One of the most argued about methodological choices in LCA is allocation, especially in multiple-output systems where allocation is needed between by-products (see e.g. Azapagic & Clift, 1999). Many studies have indicated that allocation can substantially influence the results, and the choice of allocation method usually has an ambiguous basis and, thus damages the credibility of LCA (see e.g. Curran, 2007; Reap et al., 2008). Allocation decisions are easily influenced by the perspectives and worldview of the analyst, and thus there are arguments as to whether, for instance, economic, biophysical or mass-based allocations are suitable in different case studies and decision-making situations. There are several ways to make allocations and different practices exist among LCA studies and also among LCA guidelines. While many of the recent LCA studies address multiple-output situations and emphasize the importance of the chosen allocation method, the choice of methods raises continuous debate within the research community. Guidelines, such as PAS2050 (PAS2050:2011, 2011) and the GHG-protocol (WRI/WBCSD, 2011), aim at providing a more unambiguous basis for LCA studies, also providing rules to solve multiple-output system situations using allocation. While such guidelines have been available for a while, only a few critical reviews have been published that estimate the success rate of LCA guidelines (see e.g. Ekvall & Finnveden, 2001). To our knowledge there have not been any reviews specifically targeting guideline allocation rules. The main objective here is to review critically the allocation rules and study the differences in recommendations of existing LCA guidelines for multiple-output situations, and, as an outcome, the possible differences in the final LCA results.

We chose Finnish rainbow trout fillet as our case product to study the existing LCA guidelines for multiple-output situations. The choice was made for three important reasons. Firstly, Finnish rainbow trout is the most cultivated food fish in Finland. In Finland, rainbow trout cultivation began to increase in the early 1980s and currently the production volume is around 12 million kilos per year, representing around 89 % of the total volume of fish cultivated in Finland (FGFRI, 2012). Secondly, in Finnish circumstances, the main environmental problem caused by fish cultivation is aquatic eutrophication. It is estimated that in Finland around 2 % of total aquatic phosphorous emissions and 1 % of total aquatic nitrogen emissions come from rainbow trout cultivation (Grönroos et al., 2006). In addition, climate impacts of rainbow trout are also an interesting subject of study because rainbow trout is considered to be a substitute for red meat, which is known for its high carbon footprint. Lastly, rainbow trout is an interesting case product because the production chain of rainbow trout has several multiple-output situations, for instance, between fish fillet and by-products of gutting and filleting. The most common multiple-output situations in seafood LCAs are between target fish and by-catch (capture fisheries), feed ingredients, by-products such as roe and by-products of gutting and filleting when the fish is processed.

During the past ten years there have been several LCA studies on seafood products and the methodologies for solving the multiple-output situations have varied. For instance, Ziegler et al. (2003) used economic allocation for cod. Meanwhile, Eyjólfsdóttir et al. (2003) allocated all environmental loads to filleting in fish processing in the LCA of cod. Winther et al. (2009) used mass allocation for seafood and Ayer & Tyedmers (2009) used system expansion for salmon, where fish processing waste replaced fertilizers. Allocation choices need to be made also between feed ingredients because many of the ingredients come from multiple-output situations. The choices for solving these multiple-output situations have varied among studies: Papatryphon et al. (2004) used economic allocation, Winther et al. (2009) chose mass allocation, and Ayer & Tyedmers (2009) used gross energy content. In general, the most used allocation methods for seafood LCAs are mass and economic allocation, but gross energy content and system expansion are also used in some studies (Ayer et al., 2007; Parker, 2012).

2. Methods

2.1. Functional unit, system boundaries and multiple-output situations

In this study we chose to do an attributional LCA study and the functional unit was one tonne of skinless rainbow trout fillet. The studied impact classes were climate change and aquatic eutrophication. The system boundaries included production of feed raw materials, production of feed, hatchery, farming, processing, packing, and transport (see Figure 1). Fish fillet, besides roe, is the only component for human consumption and the mass of the fillet is 52 % of the initial weight of the fish. The by-products of the process come from gutting and filleting and from roe. The by-products of gutting and filleting are sold to feed processing plants and used further as feed for fur-farming animals. The rainbow trout feed is a mixture of fish meal and oil (mainly sprat, eel and sandeel) and vegetable raw material, mainly soybean meal.

There are several relevant multiple-output situations in the LCA of cultivated rainbow trout. We chose to focus on the four main multiple-output situations (Figure 1):

1) fish meal and oil (both components of rainbow trout feed; ratio is 1:3)
2) soybean meal and oil (soybean meal is a component of rainbow trout feed)
3) round fish (whole fish) and roe
4) fish fillet and by-products of gutting and filleting

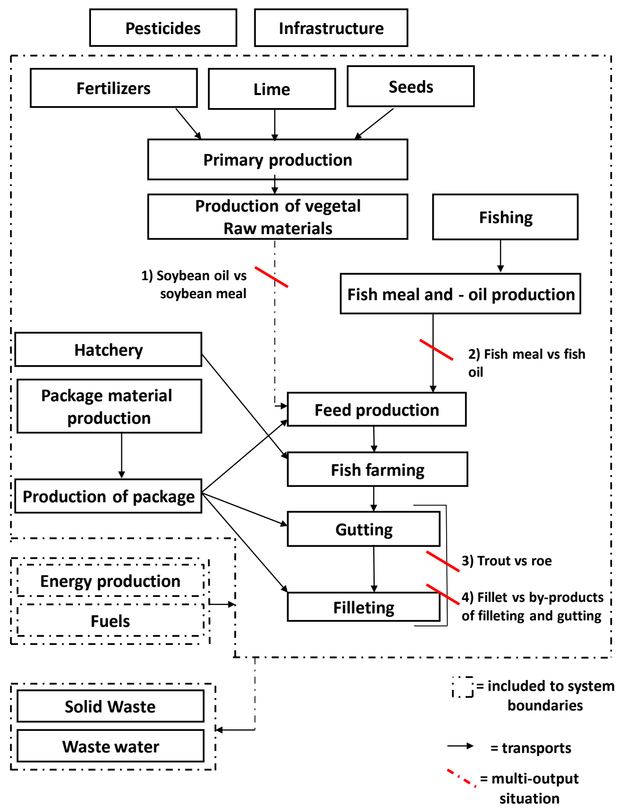

Figure 1. Multiple-output situations and system boundaries for Finnish rainbow trout.

2.2. Data sources

Rainbow trout production volumes and amounts of feed used were based on Finnish statistics and personal interviews (FGRFI, 2013; Kallioniemi, 2009, 2010). The data source of hatcheries was based on a single Finnish fish farming company (Puttonen, 2011) and the data for the nutrient emissions of the fish farm were based on official statistics (Environment Institute of South-West Finland). The fuel consumption of the boat traffic related to fish farming and electricity consumption for the fish farming were obtained from the pilot projects of the Finnish Game and Fisheries Institute (RKTL) (Kankainen et al., 2007, Setälä et al., 2009). The Plastics Europe database was used for the life cycle inventory data for polyethylene, polypropylene and polystyrene and the inventory data of the processing of the polystyrene into boxes used in fish transport. The material inputs for cultivation of one tonne of rainbow trout (round fish, before processing) are presented in Table 1.

The data for fish meal and fish oil production were based on data from a Danish producer (confidential). The modeling of the fish raw material production was based on the LCA Food-database. The inventory data for soybean cultivation were obtained from the Ecoinvent database (Jungbluth et al., 2007) and from da Silva et al. (2010).

Table 1. Energy and material inputs for production of Finnish Rainbow trout. One tonne of round fish.

Material input	
Light fuel oil, l/t	22
Electricity consumption, kWh/t	206
Nylon (Polyamid 6-6), kg/t	5.2
Polyethylene Terephthalate, kg/t	2.2
Polyethylene (HDPE), kg/t	5.8
Concrete, kg/t	13
Light fuel oil, l/t	22

2.3. Sensitivity analysis for allocation methods

In our study we built a basic case where the main four multiple-output situations were solved using allocation methods that we considered to be the most appropriate and systematic for each situation. The substitution method was not considered in the basic case because of a high level of uncertainty and because of arguments that the substitution method is not suitable for attributional LCA studies (Curran, 2007; Heijungs & Guinée, 2007). Furthermore, due to lack of practical examples of physical causality (see e.g. Schau & Magerholm Fet, 2008) we did not consider it here.

In the basic case, economic allocation was chosen for multiple-output situations between soybean oil and meal, round fish and roe, and fish fillet and by-products of gutting and filleting. In the case of soybean oil and meal, the economic value of oil is three times that of meal (FAO, 2011; Jungbluth et al., 2007) and since the products are produced for different purposes, economic allocation was chosen. In the situation between round fish and roe, roe is a very important by-product, having higher economic value (FGRFI, 2013), and thus we decided to use economic allocation. We chose to use economic allocation also between fish fillet and fish by-products of gutting and filleting because while the by-products of gutting and filleting represent around half of the weight of the fish, their economic value is very low. Furthermore, by-products of gutting and filleting would not be produced without also producing the fish fillet, and thus in our opinion more emissions should be allocated to the fillet. All in all, mass allocation was used only between fish meal and oil where mass allocation was chosen because both are of similar economic value and are used for the same purpose – as rainbow trout feed.

Besides the basic case we did a sensitivity analysis where we varied the methods to solve multiple-output situations and compared the results with those for the basic case. In addition to mass and economic allocation, the other methods used were 100-0 allocation, where all emissions were allocated to fish fillet, and a substitution method. Substitution was used between fillet and by-products of gutting and filleting. Both the processing waste from Atlantic salmon processing and captured Baltic herring from the Baltic Sea are used for fur animal feed, and thus the by-products of gutting and filleting of Finnish rainbow trout could replace both Baltic herring and the residues of Atlantic salmon processing. However, the use of Atlantic salmon processing waste as a substitute is problematic because allocation is also needed between Atlantic salmon fillet and its by-products. It should be discussed further whether the substitution method is possible if the substitute is derived using allocation. A second alternative is to assume that the by-products of rainbow trout processing replace Baltic herring, where no further allocation is needed. In this study we used the latter substitute, Baltic herring, because it does not require further allocation. What is interesting about Baltic herring is that it has negative eutrophication impact because fishing removes nutrients from the Baltic Sea.

The chosen LCA guidelines for our comparative study were ISO 14040/44 (ISO 14040, 2006; ISO 14044, 2006), ILCD-handbook (ILCD, 2010), GHG-protocol (WRI/WBCSD, 2011), PAS standard for seafood products (PAS 2050-2, 2012) and PCR Basic Module for seafood products (PCR, 2010). Only the given allocation rules of each guideline were applied.

3. Results

3.1. Basic case and sensitivity analysis

The carbon footprint of rainbow trout fillet was 4.3 t CO2-eq/t of fillet when using the basic case to solve the multiple-output situations. The sensitivity analysis showed a range of 2.3-5.2 t CO2-eq/t of fillet. The lowest result was received when only mass-based allocation was used and was highest when all was allocated to the fillet. The results of eutrophication impact were partly similar to those for carbon footprint. The eutrophication impact was 38 kg PO4-eq/t of fillet in the basic case, while when mass allocation was used, the eutrophication impact was 22 kg PO4-eq/t of fillet, and when everything was allocated to the fillet, the eutrophication impact was over 45 kg PO4-eq/t of fillet. When substitution methodology was used, the eutrophication impact was 57 kg PO4-eq/t of fillet, because fewer nutrients were removed from the Baltic Sea with captured Baltic herring. So, as a result from the sensitivity analysis, allocation had a marked effect on the final results of the LCA for Finnish rainbow trout. The sensitivity analysis showed that the most important multiple-output situations is allocation between fillet and by-products of gutting and filleting, where mass allocation halves the emissions compared with economic allocation.

3.2. Comparative study of guideline allocation rules

Overall, as shown in Table 2, the LCA results varied greatly within and among the LCA guidelines. The comparative study of allocation rules of the chosen LCA guidelines is based on our interpretation of the guidelines. To improve the transparency of results we have also explained our interpretations here.

The ISO-standard recommends substitution and that is why it was chosen when viable for multiple-output situations between fillet and by-products of gutting and filleting. The other multiple-output situations were solved using both mass and economic allocation because the standard does not recommend one over the other (ISO 14040, 2006; ISO 14044, 2006).

In applying the ILCD-handbook guidelines, substitution is not allowed in attributional LCAs. In attributional LCAs the handbook recommends avoiding allocation or solving it using physical-causal relationships, but neither was viable and therefore the third option, economic allocation, was chosen for all multiple-output situations when using the ILCD-handbook (ILCD, 2010).

The GHG Protocol substitution method was not used between fillet and the by-products of fish gutting and filleting because while the protocol recommends substitution over allocation, it states that it has to be known exactly what product is replaced and in our study it was not evident that Baltic herring was the appropriate substitute. In other multiple-output situations both mass and economic allocation were used because the Protocol does not clearly recommend one over the other (WRI/WBCSD, 2011).

In applying the PAS standard for seafood products, avoiding allocation was not a possibility, and thus the second option, mass allocation, was chosen. Furthermore, substitution was used both between fillet and the by-products of fish gutting and filleting (PAS 2050-2, 2012).

The PCR Basic Module for seafood products (PCR, 2010) recommends mass allocation. However, the choice of mass allocation was not obvious because the Supporting Annexes of PCRs state that low-value by-products should be regarded as waste and nothing should be allocated to the low-value product (EPD, 2008). However, because of the difficulties in interpreting the instructions and because of the substantial effect on the results, we decided to present both results, where emissions were first allocated to by-products of gutting and filleting based on mass, and secondly where no emissions were allocated to by-products of gutting and filleting.

Table 2. The carbon footprint (CFP) and eutrophication impact of one tonne of Finnish rainbow trout when the only variant is the allocation method based on recommendations of different LCA guidelines.

Allocation recommendation based on:	Recommended allocation method	Carbon footprint (kg CO_2-eq/ kg fillet)	Eutrophication (kg PO_4^{3}-eq/ t fillet)
ILCD-handbook	Economic	4.2	38
ISO 14040/44	Physical (mass) – Economic + Substitution (by products of gutting and filleting)	4.1 – 4.3	38 – 57
GHG Protocol	Physical (mass) – Economic	2.4 – 4.3	- (only CFP)
PAS2050-2	Mass + Substitution (by products of gutting and filleting)	4.1	- (only CFP)
PCR Basic Module for seafood products	Mass – Mass but emissions are not allocated to by-products of gutting and filleting	2.4 – 4.4	22–39

4. Discussion and conclusion

Seafood LCAs use various allocation methods and the LCA guidelines aim at providing more unambiguous rules for different multiple-output situations. In this study we did a comparative study for different LCA guidelines and interpreted the allocation rules. To our surprise it proved quite difficult to interpret guideline allocation rules even with substantial expertise – having conducted several food LCAs during past two decades. Another surprise was the variation in allocation rules within and among the current LCA guidelines. If our interpretations of the guidelines are accurate, the results vary greatly when using different recommendations, even when the only variant is choice of allocation method. Moreover, while it was not studied here, it is possible that other methodological recommendations (based on the LCA guidelines) could result in much wider variation in the results.

Application of the substitution method was also challenging. The right replacement product was not self-evident. The choice of replacement product for the by-products of gutting and filleting would have a major impact on our comparative study of guidelines. When assuming that processing waste of rainbow trout replaces Baltic herring, the eutrophication impact of fillet rises sharply because fishing of Baltic herring would actually reduce nutrients in the Baltic Sea. Overall, one of the main conclusions is that one has to be careful when using the substitution method. For instance, the GHG Protocol does not recommend substitution when there is uncertainty in defining the substitute (WRI/WBCSD, 2011).

Generally, both the sensitivity analysis and standard comparisons show that allocation has a marked effect on the final results of the LCA for Finnish rainbow trout fillet. When using mainly economic allocation, the environmental impact of a trout fillet can almost double compared with the situation where mainly mass allocation is used. One good partial solution to avoid misunderstandings and improve comparability of LCA studies is to provide arguments for the chosen allocation methods and to conduct a sensitivity analysis – presenting the results of the sensitivity analysis when communicating the final results of the LCA studies. Additionally, our study shows that because the LCA guidelines studied differ from each other they can also be interpreted differently. To reduce the ambiguity of LCA studies we suggest that more work needs to be done to improve recommendations for multiple-output situations and to harmonize the current LCA guidelines.

5. Acknowledgement

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Introduction of uncertainty into trade-offs between productivity and life cycle environmental impacts in rice production systems: Assessing the effectiveness of nitrogen-concentrated organic fertilizers

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ABSTRACT

We compared three rice production systems, the system using a pelletized nitrogen concentrated organic fertilizer, the system using a conventional organic fertilizer, and the system using a chemical fertilizer, by calculating life cycle greenhouse gas (GHG) emissions. Two-dimensional uncertainty regions were introduced into the land-oriented expression, which was established to assess efficiency of agricultural production systems on the basis of constant returns to scale (CRS) and variable returns to scale (VRS), to analyze stability of the comparison. The results indicated that the rice production system using a high nitrogen organic fertilizer was recognized as a promising alternative and that this approach was useful in understanding the stability of the results through detecting overlaps among uncertainty regions.

Keywords: uncertainty, trade-offs, organic fertilizers, yield fluctuations, methane emissions

1. Introduction

Life cycle assessment (LCA) has been applied to measure potential environmental improvements achieved by the introduction of new agricultural practices (Notarnicola et al. 2012; van der Werf et al. 2014). A typical approach is to make comparisons among several alternative agricultural production systems. For example, many comparisons have been made between organic and conventional crop production systems using LCA (Hayashi 2013; Hokazono and Hayashi 2012; Nemecek et al. 2011; Williams et al. 2010). A remarkable result in the earlier comparisons among agricultural production systems is the presence of trade-offs between environmental and economic indicators. Since improvements in area-based environmental indicators tend to entail decrease in crop yield, it is necessary to conduct an analysis of trade-offs on the two-dimensional space. In order to illustrate the implication of the space, we coined the term “land-oriented expression”, which is contrasted to “product-oriented expression” that is equivalent to a system model using the functional unit of product weight (Hayashi 2013).

The existence of hot spots in life cycle environmental impacts, however, can make the technological improvements negligible. For example, the degree of decrease in greenhouse gas (GHG) emissions from paddy rice cultivation embodied by substituting improved agricultural inputs for conventional ones can be smaller than the range of methane emission fluctuations from paddy fields.

Therefore, we extend the trade-off analysis by explicitly introducing uncertainty modeling in order to more properly assess improvement potential of agricultural technology development. In other words, uncertainty representation is introduced into the two-dimensional land-oriented expression to focus our attention to the trade-offs and to detect stability of the results.

2. Methods

2.1. An outline of the case study

The decision problem we analyzed in this study was whether to select organic fertilizers for paddy rice cultivation in Niigata Prefecture, one of the main rice production areas in Japan. Because of increased public needs to establish environmentally sustainable rice production systems, we made a research project to establish rice production systems using organic fertilizers, which are expected to resolve the problems cause by the excess of manure from livestock production. However, the application of conventional organic fertilizers makes the prediction of the response of rice plants to the application of fertilizer nitrogen difficult. Moreover, it can be a cause of lodging and yield instability.

We expected that the development of nitrogen concentrated organic fertilizers using closed composting facilities would be a promising method to resolve the problem. Therefore, we assessed the potential improvements by applying the nitrogen concentrated organic fertilizers to paddy rice production systems using life cycle assessment.

2.2. System description

We compared three rice production systems; the system using a pelletized nitrogen concentrated (high nitrogen) organic fertilizer made from poultry manure through the use of closed-air composting techniques, the system using a conventional (low nitrogen) organic fertilizer made from poultry manure using open-air composting techniques, and the systems using a chemical compound fertilizer.

This study is an ex-ante assessment because there are no facilities for making the high nitrogen organic fertilizer in Niigata Prefecture and thus we have constructed rice production scenarios to use the organic fertilizer. High and low nitrogen fertilizers prepared at two commercial companies in Mie Prefecture were used for field experiments at Niigata Agricultural Research Institute.

Furthermore, we analyzed the effects of system expansion because the use of manure necessitates considering the application of inventory models with substitution (avoided burden) (De Vries et al. 2012; Hamelin et al. 2011; Lopez-Ridaura et al. 2009; Martinez-Blanco et al. 2011a; Martinez-Blanco et al. 2010; Martinez-Blanco et al. 2009; Martinez-Blanco et al. 2011b; Prapaspongsa et al. 2010).

2.3. Analytical framework

There are two important factors in calculating life cycle GHG emissions from paddy rice production. One is direct emissions of methane from paddy fields and the other is crop yields. Earlier applications of LCA to rice illustrates that more than half of life cycle GHG emissions from rice production can be attributed to methane emissions (Blengini and Busto 2009; Hokazono and Hayashi 2012). In comparing organic and conventional rice production systems, for example, life cycle GHG emissions per product unit for each production system are highly dependent on crop yields (Hayashi 2013). In other words, even if life cycle GHG emissions per area unit from organic production are smaller than those from conventional production, life cycle GHG emissions per product unit from organic production tend to become larger than those from conventional production. The importance of these two factors implies that uncertainties due to the two factors are also high.

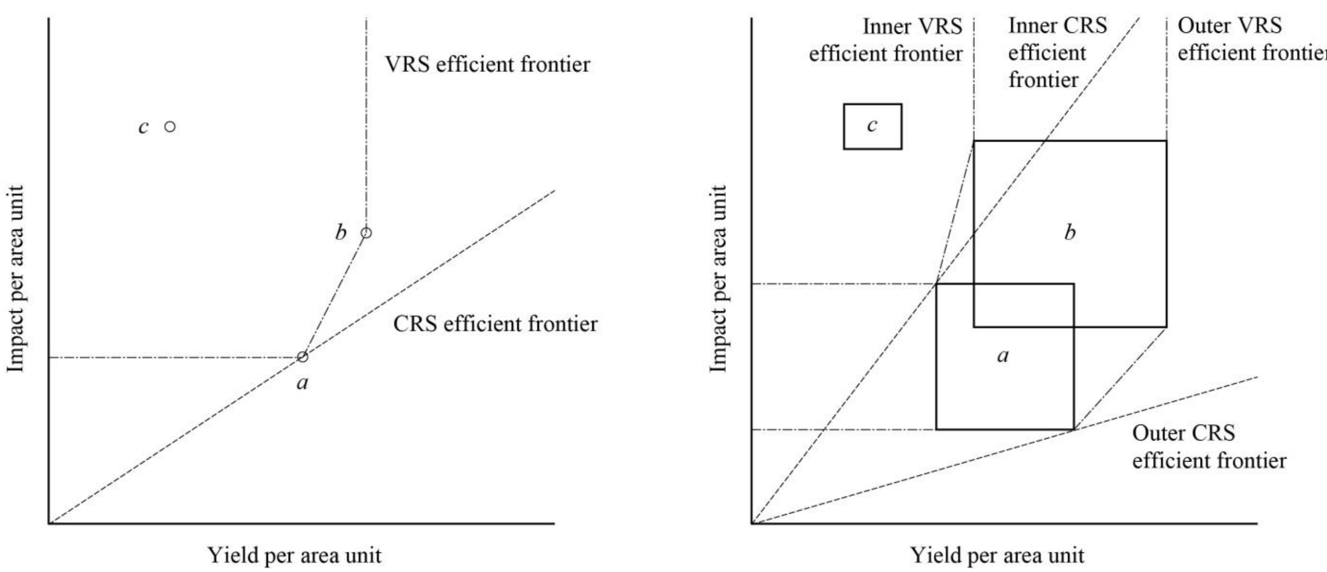

Figure 1. Two efficiency concepts defined on land-oriented expressions with and without uncertainty regions. (*a*, *b*, and *c* indicate the respective production systems.)

We apply the land-oriented expression (Hayashi 2013) to analyze the two factors. That is, using the two-dimensional space, in which the horizontal axis measures yield levels and the vertical axis measures the levels of life cycle GHG emissions, we explicitly depict uncertainties in the two factors. The land-oriented expression has an additional advantage to illustrate two efficient concepts, although the purpose of this study is the comparison and not the final selection (Figure 1). One is a concept based on constant returns to scale (CRS) and forms a straight efficient frontier, which is equivalent to the case of the product-oriented expression. The other is based on variable returns to scale (VRS) and forms a piece-wise linear efficient frontier.

In this study, we extend the expression by introducing two-dimensional uncertainty, which is depicted as an uncertainty region. As a result, outer and inner efficient frontiers are defined. In this case, the difference in efficiency between the production system a and b is somewhat inconclusive, although both a and b are more efficient than c under CRS and VRS technologies. More precisely, the efficiency orders are summarized as follows:

- the efficiency order under the CRS technology without considering uncertainty: $a \succ b \succ c$,
- the efficiency order under the VRS technology without considering uncertainty: $a \sim b \succ c$,
- the efficiency order under the CRS technology with considering uncertainty: $a \sim b \succ c$, and
- the efficiency order under the VRS technology with considering uncertainty: $a \sim b \succ c$.

2.4. Inventory data

The foreground process data were prepared on the basis of multi-year field experiments. The Japan Agricultural Life Cycle Assessment (JACLA) database (Hayashi et al. 2012) was used for background processes such as agricultural inputs including fertilizers, pesticides, and farm machines.

Data for rice cultivation are based on conventional rice production systems defined by Niigata Prefecture (Maruyama et al. 2009; Niigata Prefecture 2011). Crop yield data are gathered from field experiments at Niigata Agricultural Research Institute from 2009 to 2013 (from 2010 to 2013 for cultivation using the low nitrogen organic fertilizer). The yields are estimated using unit acreage sampling with three repetitions (Niigata Prefecture 2011).

Inventory data for organic fertilizers were prepared using experimental results conducted at Mie Prefecture Agricultural Research Institute. GHG emissions from composting processes were estimated on the basis of material balances. Emission factors for methane and dinitrogen monoxide in Greenhouse Gas Inventory Office of Japan (GIO) (2009) were used. Construction data for composting and pelletizing machines were gathered through surveys at companies.

Energy use in burning poultry manure, which was used for system expansion, was based on data available from a company in Miyazaki Prefecture. Transportation was not considered because system expansion was applied to get approximate effects of burning instead of composting.

Data on direct methane emissions from paddy fields were prepared using the results of field experiments at Niigata Agricultural Research Institute from 2009 to 2013 (from 2010 to 2013 for cultivation using the low nitrogen organic fertilizer). Measurement of methane flux was based on Minami and Yagi (1988).

2.5. Impact assessment

The impact category of global warming (IPCC 2006, 100 years) was used in impact assessment. Although we think that the other environmental impact categories such as eutrophication and acidification are important in assessing agricultural production systems with special attention to organic fertilizers, we used the category as an initial step.

3. Results

The results are illustrated in Figure 2. Horizontal ranges of the uncertainty regions show uncertainty intervals derived from annual fluctuations of crop yields. Vertical ranges are uncertainty intervals of GHG emissions derived from annual fluctuations of methane emissions. We present the results on overlaps among uncertainty regions, CRS and VRS efficiency, and the system expansion in sequence.

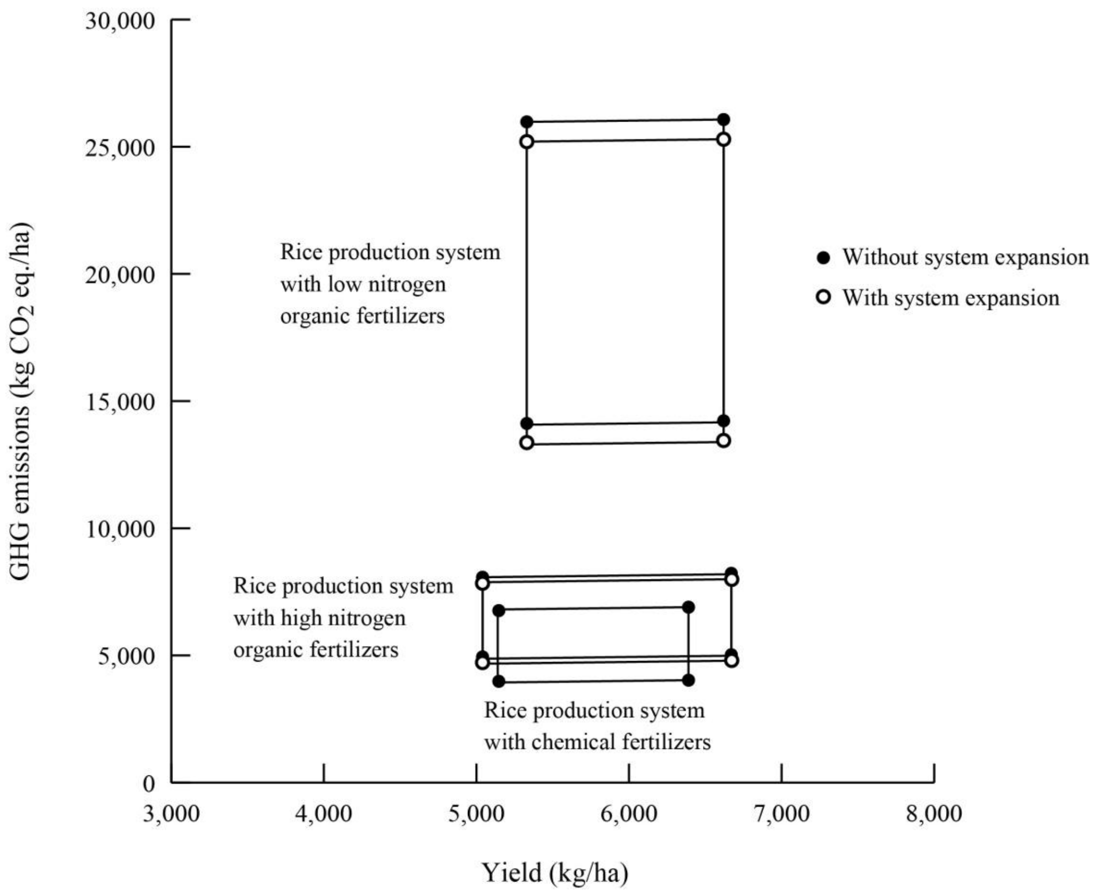

Figure 2. Uncertainty regions for each rice production system

3.1. Overlap among uncertainty regions

The configuration of two-dimensional uncertainty regions revealed that there is an overlap between the production system using the high nitrogen organic fertilizer and the system using the chemical fertilizer and that no overlap between the two systems and the system using the low nitrogen organic fertilizer.

3.2. CRS efficiency

Lines through the origin separated the following two groups: (1) the production system using the chemical fertilizer and one using the high nitrogen organic fertilizer and (2) the production system using the low nitrogen organic fertilizer. It implies that the former was CRS efficient. In other words, life cycle GHG emissions per kg of rice (as a functional unit) from the former were smaller than those from the latter.

3.3. VRS efficiency

The uncertainty region for the rice production system with the low nitrogen organic fertilizer was contained in the region between outer and inner VRS efficient frontiers. Therefore, if we use the concept of strict efficiency, we can exclude it from the systems to be selected without using preferences (weighting).

3.4. System expansion

The graphical expression also showed that the effects of systems expansion were negligible as compared with the size of the uncertainty regions. The results presented already were applicable to the both cases with and without system expansion.

4. Discussion

In summarizing these results, it is pointed out that the use of the high nitrogen organic fertilizer is promising and that the two-dimensional expressions improve our understanding about stability of results.

However, there are some limitations. The first limitation is the method to determine upper and lower bounds in intervals. Because of arbitrariness in the determination, there may be some ambiguities in judgments based on intervals. Further studies are necessary for this direction, because the problem of arbitrariness will be found in uncertainty analysis using the pedigree matrix, which is implemented in LCA software such as SimaPro and commonly used in LCA.

The second limitation is related to field measurement techniques, which are a source of arbitrariness mentioned already. Although we limited our attention to uncertainty due to annual fluctuations, uncertainty due to measurement techniques is important in agronomy and should be included into uncertainty analysis in the next step of the study.

5. Conclusion

This paper demonstrated that a method to introduce interval-based uncertainty into the two-dimensional space to detect trade-offs between productivity and GHG emissions was useful in comparing several agricultural production systems and in understanding stability of the results. The approach used in this study can be modified into the framework of land use transformation. In this case, the topic is transition from a system with chemical fertilizers to a system with organic fertilizers. An important implication of this study is that the conclusion is not straightforward in the sense that the latter is efficient as compared with the former only if organic fertilizers are made efficiently. That is, transition to a bio-based system will be established only if efficient bio-material transformation techniques are fully developed.

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Use of fertilizing residues by agricultural activities in LCA studies

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ABSTRACT

This work is a review of Life Cycle Assessment (LCA) studies dealing with agricultural use of fertilizing residues (FR). The majority of the studies were dedicated to LCA of waste and wastewater treatment systems and, in few cases, to agricultural productions. In most of the studies on LCA of waste and wastewater treatment, FR spreading induces toxicity due to heavy metals and global warming and atmospheric pollution because of emissions of nitrogen compounds. When a livestock is studied, FR spreading is generally a minor contributor to the impact compared to livestock building and animal manure stocks. The fertilizing effect of FR is taken into account by substitution of mineral fertilizers. Substitution of mineral fertilizers is the main driver of the environmental assessment result. Unfortunately, the substitution is not always explained or presented in the different studies, which makes interpretation of the results difficult. This variability of system boundaries also affects the results.

Keywords: Field application, Residues, Waste treatment, Wastewater treatment, Biosolids.

1. Introduction

Human waste and animal manures have always been used for agricultural production, but intensive farming and the increase of wastewater treatment capacity put this issue forward. Severe environmental burdens are due to intensive manure production (Mallin and Cahoon 2003), and toxicity of heavy metals contained in biosolids are often highlighted (Benítez et al. 2001).

Agricultural use of fertilizing residues (FR) is represented in several ways in Life Cycle Assessment (LCA) studies. This work is a review of LCA studies dealing with agricultural use of FR to show current LCA practices (key parameters, modeling choices for emissions and substitution) and main results of theses works. The review is presented per FR type: biosolids, organic part of municipal wastes, animal manures, digestates, and biochars.

2. Methods

The collection of articles for this review has been carried out in two steps. First, two queries have been made on the Web of Science (Thomson Reuters 2014) and CAB Abstracts (Cabi 2014) :

- Query dedicated to FR terms: "*waste*" or "residu*" or "sludge*" or "sewage*" or "biosolid*" or "*compost*" or "digestate*" or "anaerobic digest*" or "manure*" or "slurr*" or "effluent*" or "sediment" or "ash*" or "biochar" or "struvite" or "dredg*" or "by-product*" or "by product*" or "abattoir" or "dairy" or "whey" or "bone" or "ossein" or "feather" or "exogenous organic matter" or "organic amendment*".
- Query dedicated to LCA terms: "life cycle analysis" or "life cycle assessment" or "LCA" or "life cycle management" or "LCI" or "life cycle inventor*" or "impact assessment".

These queries crossed title and abstract fields in the database (FR terms in title field and LCA terms in abstract field, and vice versa) and yielded in 2229 references. To reduce the size of this set, a manual selection has been performed by scanning the abstracts to remove references

- not dealing with LCA,
- dealing with waste treatment without agricultural use (*e.g.* biosolid incineration, organic waste landfilling),
- dealing with agricultural production without FR use.

One hundred references have been selected with this approach, which are included in this study.

3. Results

The main part of the article collection is composed of recent articles from the last decade, with an overall time frame from 1998 to 2013. Figure 1 presents a word cloud built from titles of the articles. It shows that the collected articles mainly focused on waste and associated treatment (e.g. words like “waste”, “wastewater”, “municipal”, “solid”, “treatment” and “composting”). The word cloud underlines also the aim of LCA as environmental assessment of the system (“environmental”, “systems”, “greenhouse”, “emissions”, “evaluation”, “study” …). Agricultural words (“organic”, “production”, “application” and “farming”) appear, but with low occurrences.

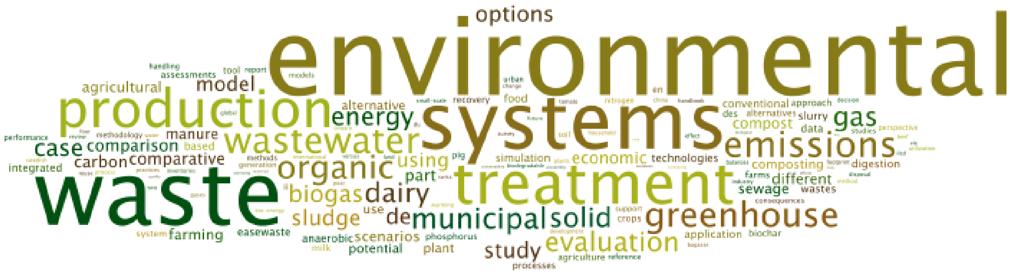

Figure 1. Word cloud from titles of the articles (words of the LCA query have been removed for better readability); The cloud was generated using the Wordle website (www.wordle.net).

4. Discussion

4.1. Overview

Few publications are dedicated only to FR uses. The main part of the literature is composed of environmental assessment of waste treatment, where agricultural use of FR is only a part of the studied system. Table 1 gives an overview of the important parameters of these publications.

When a system is multifunctional (several products or several services), substitution is often performed (avoided impacts due to avoided product). Consequently, in many cases, an agricultural use of FR is represented by an avoided mineral fertilizer production and use (41/45 references in Table 1). This allows for avoiding impacts due to production (resource consumption, plant pollution, transport …) and emissions after crop fertilization (trace metals to soil, nitrogen compounds emissions).

The key parameter for substitution of mineral fertilizers is the fertilizing effect of the FR, expressed by a Mineral Fertilize Equivalent (MFE) coefficient. But this parameter is often missing (29 references in Table 1). Noted by Heijungs and Guinée (2007) and underlined by Winkler and Bilitewski (2007), one of the main reasons for the variability of LCA results for waste treatment systems is the way how avoided impacts (due to energy or secondary matter productions) are taken into account in the system. Brockmann et al. (2014) provide a simple excel tools for MFE computation.

Emissions from spreading are dealt in three ways: (1) emissions are not taken into account (7 publications) or only partially (e.g. only heavy metals, 4 publications), (2) FR use emissions are represented, but avoided emissions not (13 publications), and (3) the difference between FR use and avoided mineral fertilizer use emissions is applied (4 publications).

4.2. Biosolids

About twenty articles deal with agricultural use of biosolids. Almost all consider the fertilizer role by substitution, but, unfortunately, the MFE value is sometimes missing, described in an internal report or it is referred to an author’s country study. When MFE values are provided (see Table 1), an important variability is observed for nitrogen equivalence, while phosphorus and potassium values are close to one.

Except for some studies (Dennison et al. 1998; Murray et al. 2008), heavy metals emissions to the environment are represented. Several authors underline the importance of this phenomenon and the induced toxicity (Hospido et al. 2004; Houillon and Jolliet 2005; Johansson et al. 2008; Wenzel et al. 2008).

Hospido and coauthors (Hospido et al. 2010) take into account thirteen drug residues in the toxicity assessment through a new characterization factor for the CML method (Muñoz et al. 2009). In this work and with this approach, the authors conclude that the drug residues’ impact is negligible compared to the heavy metals burden.

Table 1. Main characteristics of waste and wastewater treatment LCA studies.

References	**FR type**[a]	**Impacts**[b]	**Fertilizer substitution**	**Mineral Fertilizer Equivalent**	**Carbon storage / organic matter**	**Field emission (FR use)**	**Field emission substitution (avoided fertilizer)**
(Dennison et al. 1998)	USS	ND	No	NA	No	ND	No
(Murray et al. 2008)	USS	cc + inventory	Yes (N, P)	N: 1 / P: 1	No	No	No
(Hospido et al. 2004)	USS	CML	Yes (N, P)	ND	No	Yes (HM)	No
(Houillon and Jolliet 2005)	USS	eb; cc	Yes (N, P, K, lime)	N: 0.61 / P: 0.7 / K ND	No	No	No
(Wenzel et al. 2008)	USS	EDIP	ND	ND	No	ND	ND
(Johansson et al. 2008)	USS – CUSS	cc, acid. eut	Yes (N, P)	N: USS 0.42; CUSS 0.3 / P: USS 0.7 CUSS 0.035	No	Yes (N_2O, CH_4, NH_3, NO_3^-)	Yes (N_2O, CH_4, NH_3, NO_3^-)
(Beavis and Lundie 2003)	ISS	eut, cc, ph ox, acid, ecotox	Yes (N, P)	Yes (Substituted value)	No	ND	No
(Hospido et al. 2005)	USS	CML	Yes (N, P)	ND	No	Yes (CH_4, HM)	No
(Tarantini et al. 2007)	CUSS	CLM+ tox	No	NA	No	Yes (HM)	No
(Peters and Rowley 2009)	CUSS	eeb, cc, tox, ecotox	Yes (N, P, lime)	Yes (Substituted value)	Yes	Yes (HM)	No
(Hong et al. 2009)	CUSS	cc, acid, tox, lu	Yes (ND)	ND	No	Yes (HM)	No
(Liu et al. 2013)	USS	cc	Yes (N, P)	ND	No	No	No
(McDevitt et al. 2013)	USS	CML + USE-TOX	Yes (ND)	ND	No	Yes (N_2O, leaching)	No
(Lundin et al. 2000)	USS	inventory	Yes (N, P)	N: 0.5 / P: 0.7 – 1	No	No	No
(Pasqualino et al. 2009)	USS	CML	Yes (N, P)	ND	No	Yes (ND)	No
(Foley et al. 2010)	USS	inventory	Yes (N, P)	N: 0.25 – 0.75 / P: 0.25 – 0.75	Yes	Yes (N_2O, NH_3, HM)	Yes (N_2O, NH_3, HM)
(Hospido et al. 2010)	USS	CML	Yes (N, P)	N 0.5 / P 0.7	No	Yes (N_2O, NH_3, PO_4^{3-}, HM, TOC)	No
(Sablayrolles et al. 2010)	USS	acid, eut, cc, oz dep, ph ox, ecotox, tox	Yes (N, P)	Yes (Substituted value)	No	Yes (N_2O, HM, TOC)	Yes (N_2O, HM, TOC)
ORWARE[c]	USS – COW	ph ox, eut, acid, cc	Yes (N, P)	ND	No	Yes (N_2O, NH_3, NO_3^-, NO_x, P, HM)	No
EASYWASTE[d]	COW	EDIP	Yes (N, P, K,)	ND	Yes	Yes (N, P, ETM, TOC)	No
(Jury et al. 2010)	DCE	ECOINDICATOR 99, cc, eb	No (allocation)	Yes (N: 0.54 – 0.83 / P: ND / K: ND)	No	Yes (N_2O, NH_3, NO_x, NO_3^-, PO_4^{3-}-)	NA
(De Vries et al. 2012)	DPS	RECIPE (m)	Yes (N, P, K)	N: 0.65 / P: 1 / K: 1	Yes	Yes (CH_4, NH_3, NO, N_2O, NO_3^-, PO_4^{3-})	No
(Poeschl et al. 2012b; Poeschl et al. 2012a)	DC	RECIPE (m, e)	Yes (N, P, K)	N: DFB 0.45; DC 0.65 / P: 1 / K: 1	No	Yes (CH_4, NH_3, NO, N_2O, NO_3^-, PO_4^{3-})	No
(Hamelin et al. 2011)	DPS - DCS	EDIP	Yes (N, P, K)	N: PS 0.7; CS 0.75; CM 0.85/ P: 0.81 / K: 0.97	Yes	Yes (CH_4, NH_3, NO, N_2O, NO_3^-, PO_4^{3-})	No

Yes: included; No: not included or not presented; NA: non applicable; ND: not documented; HM: heavy metals; TOC: trace organics compounds

[a] CM: cattle manure; CS: cattle slurry; PS: pig slurry; USS: urban sewage sludge; CUSS: compost of urban sewage sludge; ISS: industrial sewage sludge; COW: compost of organic waste; DEC: digestate of energetic crop production; DC: digestate from codigestion (animal manures, agricultural residues, dedicated crop productions…); DPS: digestate of pig slurry; DCM: digestate from cattle manure; LF: leachate (liquid fraction) separated from digestate.

[b] Impact assessment method: CML, ECOINDICATOR 99, EDIP, RECIPE (*m* midpoint ; *e* endpoint), USETOX (ILCD handbook (European Commission - Joint Research Centre - Institute for Environment and Sustainability 2010) lists and describes the impact assessment method, see this document for details) / impacts of a method: *cc* climate change ; *oz dep*: ozone depletion; *eb* energetic balance : *acid* acidification ; *eut* eutrophication ; *ph ox* photochemical oxidation; *tox* human toxicity; *ecotox* ecotoxicity ; *lu* land uses / *inventory*: references without impact assessment.

[c] ORWARE model references: USS: (Kärrman and Jönsson 2001; Lundin et al. 2004; Tidåker et al. 2006) – COW: (Dalemo et al. 1997; Sonesson et al. 1997; Dalemo et al. 1998; Thomsson 1999; Sonesson et al. 2000; Eriksson et al. 2002; Mendes et al. 2003; Eriksson et al. 2005)

[d] EASEWASTE model references: (Kirkeby et al. 2006a; Kirkeby et al. 2006b; Christensen et al. 2007; Bhander et al. 2008; Boldrin and Thomas-Hojlund 2008; Bhander et al. 2010; Boldrin et al. 2010; Bernstad and la Cour Jansen 2011; Manfredi et al. 2011; Andersen et al. 2012)

Some works do not deal with nitrogen and phosphorus emissions to the environment due to FR spreading (Beavis and Lundie 2003; Hospido et al. 2005; Houillon and Jolliet 2005; Tarantini et al. 2007; Murray et al. 2008; Hong et al. 2009; Peters and Rowley 2009; Liu et al. 2013; McDevitt et al. 2013) and deal only with the benefit of avoided fertilizer. Other works present agricultural field emissions without description (Pasqualino et al. 2009), according to the ORWARE model (Kärrman and Jönsson 2001; Lundin et al. 2004; Tidåker et al. 2006), partially without nitrate leaching (Lundin et al. 2000), according to experimental data (Johansson et al. 2008) or determined according to models (Nemececk and Schnetzer 2012) proposed by Ecoinvent guidelines (Foley et al. 2010; Hospido et al. 2010). To finish, few works (Sablayrolles et al. 2010) consider both FR emissions and avoided emissions due to avoided mineral fertilizer use. Recently, Yoshida et al. (2013) reviewed 35 LCA studies dedicated to sewage sludge treatment (28 with agricultural use). This work shows the variability of the perimeters: 11 references with emissions to air during biosolids spreading, 11 with heavy metals emissions to soil, and 2 with carbon storage. A substitution of mineral fertilizer is done in 25 of them.

4.3. Organic fraction of municipal wastes

The use of LCA to assess waste treatment scenarios leads to dedicated simulation models. About 10 models can be found (Gentil et al. 2010), but only two have been used and take into account FR use.

ORWARE (ORganic WAste REsearch) (Dalemo et al. 1997; Eriksson et al. 2002) is a Swedish model, which has been used for real cases of waste management (Sonesson et al. 1997; Sonesson et al. 2000; Mendes et al. 2003; Eriksson et al. 2005). The model focuses mainly on climate change, acidification and eutrophication; toxicity is not addressed. FR spreading is represented with a substitution and field emissions: a simplified model of the nitrogen cycle allows determining nitrogen emissions (N_2O, NH_3, NO_3^-) and the nitrogen content replacing mineral nitrogen in the first year and the long-term. Ammonia losses are determined by Swedish experimental values, nitrate leaching by a simulation model with soil features, and nitrous oxide emissions according to IPPC recommendations. Phosphorus dynamics are assumed analogous for FR and mineral fertilizer. Dalemo et al. (1998) discuss the importance of nitrogen emissions due to FR use for the assessment of waste management scenarios. Thomsson (1999) uses ORWARE and mentions heavy metals, but without assessment because of complexity of the soil-plant mass balance.

EASEWASTE (Environmental Assessment of Solid Waste Systems and Technologies) (Kirkeby et al. 2006a; Bhander et al. 2008; Bhander et al. 2010) is a Danish model and is probably the most used LCA waste treatment tool. Assessment is carried out with the EDIP method (Wenzel et al. 1997). The model includes mineral fertilizer substitution and nitrogen, phosphorus, heavy metals and trace organic compounds emissions. Carbon storage is also modeled. However, the FR use module of the EASEWASTE model is only described in a Danish document (Hansen 2004). Effects of FR use on soil quality are not represented because of the complexity of the relation. This model shows the consequences of heavy metal emissions from agricultural residues (Kirkeby et al. 2006b) and underlines the major role of final waste destination to waste treatment assessment (Christensen et al. 2007).

EASEWASTE has been used to compare incineration, landfilling and composting of individual waste fractions (Manfredi et al. 2011). The authors observe a benefit for the composting scenario on ecotoxicity because of avoided emissions of mineral fertilizer production (chromium and mercury). EASEWASTE has been used to assess commercial compost from food waste and garden waste, which is used as growth media preparation instead of peat (Boldrin and Thomas-Hojlund 2008; Boldrin et al. 2010). The benefit of this compost on climate change (because of biogenic carbon) and eutrophication (avoided nitrogen emissions of mineral fertilizer production) is shown, but it is tempered by heavy metals emissions to soil.

Some authors (Bernstad and la Cour Jansen 2011; Andersen et al. 2012) represent the organic matter supply effect of the use of compost from household food waste by a peat substitution (substitution according to volume, 1 m^3 of peat for 1 m^3 of compost). They consider that the environmental burden is mainly determined by greenhouse gases emissions from the composting process and the benefit from the substitution (climate change, eutrophication, toxicity and ecotoxicity). Andersen et al. (2010) investigated substitution practices for compost use in gardening in two Danish towns (Aarhus and Copenhagen). This work shows a substitution only for 22 and 24% of the situations for peat, 12 and 24% for fertilizer, and 7 and 15% for manure.

Bernstad and la Cour Jansen (2012) reviewed LCAs for food waste management, mainly with respect to climate change. They reveal a large variability of the results and that the use of compost and digestate is profitable from an environmental point of view in some works, but negligible in other ones. The variability of the results is explained by the MFE value, the environmental impact of substituted fertilizer and carbon storage. The authors advise to consider nitrogen emissions from fertilizers in accordance with international recommendations (as ILCD) for impact assessment. But variability and lack of knowledge are again underlined. Only one reference with FR and avoided emissions is cited (Møller et al. 2009).

While most publications deal with waste treatment scenarios, Martinez Blanco et al. (2009) compare mineral fertilization to the use of compost from municipal organic waste for tomato crops. In this case, the use of compost avoids landfilling and landfill impacts are subtracted to the system: this substitution drives the environmental impacts and sets compost as best solution. A similar work, yielding in the same conclusions has been done for compost from wine shoot and sewage sludge (Ruggieri et al. 2009).

4.4. Animal manures

About twenty articles dedicated to cattle and pig productions have been found. Manure and slurry spreading is commonly represented. FR use impacts are usually negligible in comparison to emissions from livestock building and animal manure storage (Beauchemin et al. 2010; O'Brien et al. 2011) (and also for poultry production (Leinonen et al. 2012)). Animal manure storage and spreading are also often merged in a single step (Sonesson 2005; Thomassen et al. 2009).

First works on livestock LCA (Cederberg and Mattsson 2000; Cederberg and Flysjo 2004; Sonesson 2005) used a Swedish model for nitrate leaching (Aronsson and Torstensson 2003). Recent works (Fantin et al. 2012; Jan et al. 2012) follow EcoInvent guidelines (Nemecek and Kägi 2007; Nemececk and Schnetzer 2012) for all emissions. When EcoInvent is not used, emissions are usually computed by national references and models (Anton et al. 2005; Antón et al. 2005; Cooper et al. 2011; Leinonen et al. 2012; Uchida and Hayashi 2012). But nitrous oxide and methane emissions are determined in most of cases by IPCC guidelines (Cederberg and Mattsson 2000; Beauchemin et al. 2010; Hörtenhuber et al. 2010; Rotz et al. 2010; O'Brien et al. 2011; Yan et al. 2011; Fantin et al. 2012; Mc Geough et al. 2012; O'Brien et al. 2012; Bonesmo et al. 2013). The use of IPCC guidelines is also found for crop production studies (Mattsson and Wallen 2003; Cooper et al. 2011; Kimming et al. 2011; Nemecek et al. 2011a; Nemecek et al. 2011b; Hakala et al. 2012). Langevin et al. (2010) work on nitrogen emissions from slurry spreading. From a literature review, they show that pedoclimatic conditions imply a variability of LCA results that can be larger than the variability resulting from spreading technics.

Nemecek and coauthors (Nemecek et al. 2011b) underline the interest of animal manure use as fertilizer with regard to resource depletion (fossil and mineral) and soil quality. This benefit is tempered by nutrient leaching due to management complexity of the organic fertilization.

4.5. Digestates

Recently, several environmental studies of anaerobic digestion plants with agricultural use of the digestate have been published. Here, again, emissions are often computed according to EcoInvent guidelines (Jury et al. 2010; De Vries et al. 2012; Poeschl et al. 2012b; Poeschl et al. 2012a). Jury et al. (2010) used an allocation approach instead of substitution to represent digestate use. A monetary value of the avoided mineral fertilizer was used. Hamelin et al. (2011) compared slurry spreading to anaerobic digestion and digestate spreading. The MFE value followed Danish legislation. Nitrous oxide emissions were calculated according to IPCC guidelines, and other nitrogen emissions were determined with a Danish model. Carbon storage was represented according to (Petersen et al. 2002). This work incorporates crop yield variations according to fertilizer type (mineralization process of the anaerobic digestion step; the nitrogen of digestate is more available for plants than nitrogen from slurry; an increase of 9 kg of wheat per kg of nitrogen was considered). Crop yield variations are represented by avoided wheat production. However, the assessments are mainly driven by avoided fertilizer, with the yield increase effect being negligible.

4.6. Biochars

Recent works focused on agricultural use of biochar (Kameyama et al. 2010; Roberts et al. 2010; Hammond et al. 2011; Ahmed et al. 2012; Mattila et al. 2012; Cao and Pawlowski 2013; Sparrevik et al. 2013). Carbon storage is usually the driver of the study and biochar use appears as an interesting solution. Cao and Pawlowski (2013) underline also the benefit on climate change and the energetic balance because of avoided mineral fertilization (They assume that biochar soil-enrichment implies a decrease of 10% of mineral nitrogen, phosphorus and potassium fertilizer use because of better nutrient bioavailability).

5. Conclusion

Agricultural use of residues is a common way for waste valorization and the fertilizing function is often represented in LCA of waste treatment systems. The use of residues as fertilizer induces usually significant impacts on the ecosystems due to heavy metals and on climate change and atmospheric pollution because of nitrogen compounds emissions. The benefit from the avoided mineral fertilizer production can counter balance the impacts from spreading emissions, but, unfortunately, this substitution is often misdescribed or only partially documented. Because MFE is a key parameter for LCA of agricultural use of residues, it should be clearly presented in the studies.

Field emissions of FR differ from mineral fertilizer emissions because of management practices and nitrogen forms. In LCA works, the variety of the system limit (no emissions, FR emission only or FR and mineral fertilizer balance) implies a variability of the results, which should be integrated in the interpretation of the results.

An LCA study dealing with agricultural use of residues should be carried out notably with:

- the references, the rules and/or the models and their parameters used to determine field emissions,
- the value used as MFE, supplemented with references and sensitivity analysis,
- a description of the system limit, which has to include the whole substituted system (mineral fertilizer emission from the production to the field emission).

Some aspects are not represented in LCA. Pathogens and health consequences are not assessed in LCA, which has to be highlighted for human waste and animal manure uses. LCA works start to deal with this (Motoshita et al. 2010), but methods are not operational. Even if methods have been developed (Milà i Canals et al. 2007; Garrigues et al. 2012), the effects of the FR organic matter on soil quality is not taken into account. Recent publications deal with carbon storage and the consequences on climate change (this can be observed in publications dedicated to biochar, for example). This should be generalized in the next years.

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Improvement of the characterization factor for biotic-resource depletion of fisheries

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ABSTRACT

Langlois et al. (2012; 2014a) proposed characterization factors (CF) for fish biotic resource extraction impact assessment at the species level. This paper is an improvement of this approach. In the present work, the CF depends on the Maximum Sustainability Yield (MSY), weighted by the ratio of the current total fishing effort to the fishing effort at the MSY value. Because this ratio often cannot be computed from current databases, it is here obtained from the ratio of total catches to MSY and roots of the parabola linking catches to fishing effort. The new version of the CF is proposed for 125 fish stocks. This work allows assessment of fisheries in the LCA formalism. It contributes to a better representation of the depletion of biotic resources.

Keywords: Biotic resource depletion, Fisheries, Maximum sustainable yield, Characterization factor

1. Introduction

Food production is a major sector for environmental burden due to increase of foodstuff demand and food production intensification. The sea is more and more exploited for human food, and environmental assessment approaches have to take into account this use. Seafood products, coming from direct fishery activities or from aquaculture, where a part of the feeding comes from fish catches, induce a large consumption of biotic resources for human activities. An environmental assessment of these systems by Life Cycle Assessment (LCA) cannot be performed without a focus on this impact.

Langlois et al. (2014b) review the use of the sea in LCA and propose related impact pathways dedicated to the biodiversity damage potential, the ecosystem services damage potential, climate change, and the biotic resource damage potential. Recently, Emanuelsson et al. (2014) proposed new characterization factors (CFs) for fish resource depletion based on the lost potential yield (LPY). The LPY represents the average of lost catches owing to ongoing overfishing and is assessed by simplified biomass projections covering different fishing mortality scenarios. This useful approach provides CFs for 31 European fish species, but several parameters are needed: CFs are computed from (1) current biomass and fish mortality, and (2) target biomass and target fish mortality to Maximum Sustainability Yield (MSY).

The biotic resource damage potential is studied in Langlois et al. (2012; 2014a) for fishing activities at both species, for a given stock, and ecosystem levels. Proposed CF values have been used by Avadí et al. (2014). For the species level, CFs are built based on MSY and catches to deal with the biotic resource states. The present work is an improvement of CF determination at the species level.

2. Methods

2.1. Maximum Sustainability Yield.

The relation between catches and fishing effort is commonly modelled by a parabola, where the MSY is equal to the catches at the inflexion point (see Figure 1). Fish stocks are classically assessed with MSY: the highest fish catch that can be sustained in the long term (Graham 1935; Schaefer 1991). Fishery exploitation of a given stock at time t (C_t) can be increased up to a maximum level by increasing the fishing effort (E_t), because the catches are compensated by an equivalent fish production. Above the MSY and its corresponding E_{MSY}, renewal of the resource (reproduction and body growth) cannot keep pace with the removal caused by fishing and natural mortality. The MSY can be estimated either with a variety of stock-assessment methods or empirically. The most useful database for this is the RAM Legacy Stock Assessment Database (Ricard et al. 2012).

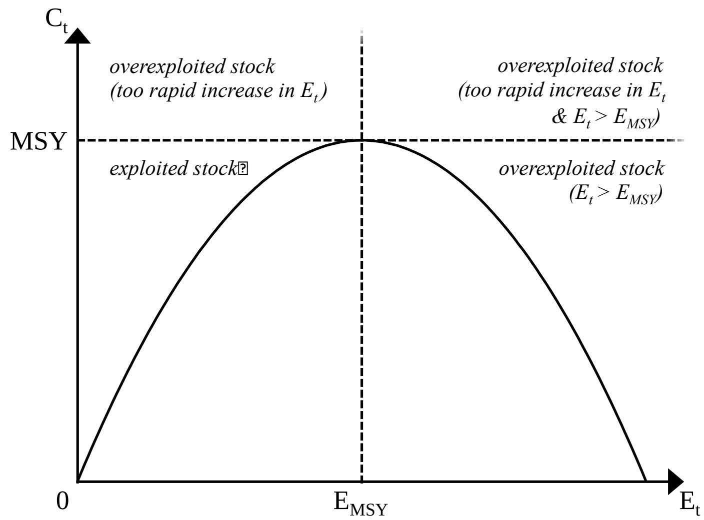

Figure 1. Fish catch (C_t) as a function of fishing effort (E_t) at the steady state

2.2. Characterization factors previously proposed

In Langlois et al. (2012; 2014a), CFs are computed as follows

$$\mathrm{CF} = \begin{cases} \frac{1}{\mathrm{MSY}} & \text{if exploited} \\ \frac{1}{\mathrm{MSY}} \times \frac{\mathrm{MSY}}{\bar{C}_t} = \frac{1}{\bar{C}_t} & \text{elseif} \end{cases} \quad \text{Eq. 1}$$

where $\bar{C}_t$ represents mean fish catches over five years prior to impact assessment to approximate the equilibrium value (if average catches are higher than the MSY due to non-equilibrium situation, $\bar{C}_t$ is set equal to MSY). For an exploited stock, CF allows to place landings in front of renewability of the stock. When the stock is overexploited, the ratio of MSY to $\bar{C}_t$ is added. The ratio varies from 1 to infinity for catch rates ranging from MSY to zero (i.e. when the stock is overexploited close to MSY or when it is severely depleted, respectively). The ratio allows introducing the seriousness of the overexploitation based on catches only, which are common data (unfortunately, fishing efforts are often unavailable data). See Langlois et al. (2014a) for the determination of the stock status (i.e. overexploited or not).

2.3. Improvement of the characterization factors

In the present work, to determine the CF, MSY is changed in accordance with the ratio $\frac{E_t}{E_{\mathrm{MSY}}}$, which describes the extent of exploitation or overexploitation. Let the relation presented in Figure 1 be

$$\bar{C}_t = -a_1 E_t^2 + a_2 E_t \quad \text{Eq. 2}$$

The derivative is the following:

$$\frac{d\bar{C}_t}{dE_t} = -2a_1 E_t + a_2 \quad \text{Eq. 3}$$

$$\left.\frac{d\bar{C}_t}{dE_t}\right|_{E_t = E_{\mathrm{MSY}}} = 0 = -2a_1 E_{\mathrm{MSY}} + a_2 \quad \text{Eq. 4}$$

With (2) and (4), we have

$$\bar{C}_t = -a_1 E_t^2 + 2a_1 E_{\mathrm{MSY}} E_t \quad \text{Eq. 5}$$

$$\mathrm{MSY} = -a_1 E_{\mathrm{MSY}}^2 + 2a_1 E_{\mathrm{MSY}} E_{\mathrm{MSY}} \quad \text{Eq. 6}$$

$$\frac{\bar{C}_t}{\mathrm{MSY}} = -\left(\frac{E_t}{E_{\mathrm{MSY}}}\right)^2 + 2\frac{E_t}{E_{\mathrm{MSY}}} \quad \text{Eq. 7}$$

The polynomial (7) allows finding the ratio of fishing effort to fishing effort for the MSY as a function of the ratio of catches to MSY. The roots are the following:

$$r_1 = 1 + \sqrt{1 - \frac{\bar{C}_t}{\mathrm{MSY}}}, r_2 = 1 - \sqrt{1 - \frac{\bar{C}_t}{\mathrm{MSY}}} \quad \text{Eq. 8}$$

and are used in the CF to weigh MSY

$$\mathrm{CF} = \begin{cases} \dfrac{1}{\mathrm{MSY}\left(1+\sqrt{1-\frac{\bar{C}_t}{\mathrm{MSY}}}\right)} & \text{if exploited} \\ \dfrac{1}{\mathrm{MSY}\left(1-\sqrt{1-\frac{\bar{C}_t}{\mathrm{MSY}}}\right)} & \text{elseif} \end{cases} \quad \text{Eq. 9}$$

This weight varies theoretically from 2 (unexploited resource) to 0 (totally depleted resource) and introduces a relative position for the stocks according to the fishing effort only with catches and MSY values.

3. Results

The RAM Legacy database includes biological reference points for over 361 stocks, of which 138 have MSY values and 313 have catches data. The number of stocks with both values is 125. Table 1 gives CF values for overexploited (55) and exploited (70) stocks, where both catches and MSY are available in the RAM legacy database. Figure 2, left part, shows the CF weight factors according to the relative finishing effort both for previous (1 for exploited, $\frac{\mathrm{MSY}}{\bar{C}_t}$ for overexploited stock) and current CFs $\left(1 \pm \sqrt{1 - \frac{\bar{C}_t}{\mathrm{MSY}}}\right)$. Figure 2, right part, shows the total biotic resource depletion impacts (i.e. $\bar{C}_t \times \mathrm{MSY}$) for both previous and current work.

Table 1. Characterization factors (CF) for biotic-resource depletion at the species-stocks level in the RAM Legacy database.

Overexploited Stock ID	Common name	CF ($kg^{-1}.y$)	Exploited Stock ID	Common name	CF ($kg^{-1}.y$)
ACADREDGOMGB	Acadian redfish	2.76×10^{-09}	ALBASPAC	Albacore tuna	1.48×10^{-11}
ALBANATL	Albacore tuna	4.80×10^{-11}	ARFLOUNDPCOAST	Arrowtooth flounder	1.00×10^{-10}
AMPL5YZ	American Plaice	9.77×10^{-10}	ARGANCHONARG	Argentine anchoita	1.20×10^{-12}
ANCHOVYKILKACS	Anchovy kilka	4.87×10^{-10}	ARGANCHOSARG	Argentine anchoita	1.73×10^{-12}
ARGHAKENARG	Argentine hake	2.09×10^{-11}	ATOOTHFISHRS	Antarctic toothfish	2.32×10^{-10}
ARGHAKESARG	Argentine hake	3.60×10^{-12}	AUSSALMONNZ	Australian salmon	1.98×10^{-10}
ATBTUNAEATL	Atlantic bluefin tuna	2.03×10^{-11}	BGROCKPCOAST	Blackgill rockfish	2.80×10^{-09}
ATBTUNAWATL	Atlantic bluefin tuna	8.03×10^{-10}	BHEADSHARATL	Bonnethead shark	8.79×10^{-13}
ATHAL5YZ	Atlantic Halibut	3.63×10^{-08}	BIGEYEIO	Bigeye tuna	8.99×10^{-12}
BIGEYEATL	Bigeye tuna	1.53×10^{-11}	BIGEYEWPO	Bigeye tuna	1.55×10^{-11}
BLACKOREOWECR	Black oreo	7.95×10^{-10}	BLACKROCKNPCOAST	Black rockfish	8.05×10^{-10}
BLUEROCKCAL	Blue rockfish	5.81×10^{-09}	BLACKROCKSPCOAST	Black rockfish	6.11×10^{-10}
BOCACCSPCOAST	Bocaccio	2.59×10^{-08}	BLUEFISHATLC	Bluefish	4.55×10^{-11}
BSBASSMATLC	Black sea bass	4.88×10^{-10}	BTIPSHARATL	Blacktip shark	3.36×10^{-11}
BUTTERGOMCHATT	Atlantic butterfish	8.21×10^{-08}	BTIPSHARGM	Blacktip shark	2.07×10^{-11}
CABEZSCAL	Cabezon	7.54×10^{-08}	CABEZNCAL	Cabezon	5.43×10^{-09}
CODGB	Atlantic cod	3.87×10^{-10}	CHAKESA	Shallow-water cape hake	6.37×10^{-12}
CODGOM	Atlantic cod	5.18×10^{-10}	CHILISPCOAST	Chilipepper	2.36×10^{-10}
CROCKPCOAST	Canary rockfish	3.83×10^{-08}	CMACKPCOAST	Pacific chub mackerel	2.65×10^{-11}
DEEPCHAKESA	Deep-water cape hake	9.38×10^{-12}	DSOLEPCOAST	Dover sole	3.50×10^{-11}
DKROCKPCOAST	Darkblotched rockfish	1.59×10^{-08}	ESOLEPCOAST	English sole	1.28×10^{-10}
GAGGM	Gag	4.46×10^{-10}	FLSOLEBSAI	Flathead sole	4.27×10^{-12}
GRAMBERGM	Greater amberjack	1.03×10^{-09}	GEMFISHNZ	common gemfish	3.86×10^{-10}

Overexploited			**Exploited**		
Stock ID	**Common name**	**CF ($kg^{-1}.y$)**	**Stock ID**	**Common name**	**CF ($kg^{-1}.y$)**
HAD5Y	Haddock	7.35×10^{-10}	GOPHERSPCOAST	Gopher rockfish	9.90×10^{-09}
KMACKGM	King Mackerel	3.91×10^{-10}	HADGB	Haddock	1.69×10^{-11}
NZSNAPNZ8	New Zealand snapper	6.48×10^{-10}	HERRNWATLC	Herring	3.14×10^{-12}
POLL5YZ	Pollock	1.94×10^{-10}	KELPGREENLINGORECOAST	Kelp greenling	7.98×10^{-09}
POPERCHPCOAST	Pacific ocean perch	1.54×10^{-08}	KINGKLIPSA	Kingklip	7.57×10^{-12}
PTOOTHFISHMI	Patagonian toothfish	5.42×10^{-09}	KMACKSATLC	King Mackerel	3.88×10^{-10}
RGROUPGM	Red grouper	2.86×10^{-10}	LNOSESKAPCOAST	Longnose skate	5.35×10^{-10}
RPORGYSATLC	Red porgy	3.20×10^{-08}	LSTHORNHPCOAST	Longspine thornyhead	1.56×10^{-10}
RSNAPEGM	Red snapper	1.08×10^{-09}	MACKGOMCHATT	Mackerel	7.85×10^{-12}
RSNAPWGM	Red snapper	8.77×10^{-10}	MONKGOMNGB	Monkfish	2.86×10^{-10}
SBARSHARATL	Sandbar shark	2.91×10^{-08}	NRSOLEEBSAI	Northern rock sole	1.72×10^{-12}
SNOWGROUPSATLC	Snowy grouper	7.04×10^{-09}	NZLINGLIN3-4	Ling	5.89×10^{-11}
SPANMACKSATLC	Spanish mackerel	7.27×10^{-10}	NZLINGLIN5-6	Ling	2.75×10^{-11}
STMARLINSWPO	Striped marlin	6.56×10^{-10}	NZLINGLIN6b	Ling	7.00×10^{-10}
STRIPEDBASSGOMCHATT	Striped bass	9.89×10^{-11}	NZLINGLIN72	Ling	1.41×10^{-09}
SWORDMED	Swordfish	8.24×10^{-11}	NZLINGLIN7WC	Ling	9.32×10^{-11}
SWORDNATL	Swordfish	1.16×10^{-10}	OROUGHYNZMEC	Orange roughy	2.37×10^{-10}
TAUTOGRI	Tautog	7.44×10^{-09}	PATGRENADIERSARG	Patagonian grenadier	7.57×10^{-12}
TILESATLC	Tilefish	6.55×10^{-09}	PHAKEPCOAST	Pacific hake	1.32×10^{-12}
VSNAPSATLC	Vermilion snapper	3.83×10^{-09}	POPERCHGA	Pacific ocean perch	3.11×10^{-11}
WHAKEGBGOM	White hake	4.87×10^{-10}	PSOLENPCOAST	Petrale sole	3.90×10^{-10}
	Windowpane	1.79×10^{-09}	PSOLESPCOAST	Petrale sole	4.06×10^{-10}
WINDOWSNEMATL	Windowpane	3.63×10^{-09}	PTOOTHFISHPEI	Patagonian toothfish	2.43×10^{-10}
WINFLOUN5Z	Winter Flounder	8.03×10^{-10}	REDFISHSPP3LN	Redfish species	2.07×10^{-11}
WINFLOUNSNEMATL	Winter Flounder	9.25×10^{-10}	SABLEFEBSAIGA	Sablefish	3.73×10^{-11}
WITFLOUN5Y	Witch Flounder	4.25×10^{-10}	SABLEFPCOAST	Sablefish	1.16×10^{-10}
WPOLLEBS	Walleye pollock	1.02×10^{-12}	SBWHITACIR	Southern blue whiting	5.19×10^{-11}
WPOLLGA	Walleye pollock	2.80×10^{-11}	SKJCWPAC	Skipjack tuna	6.43×10^{-13}
YELLCCODGOM	Yellowtail flounder	1.49×10^{-09}	SKJEATL	Skipjack tuna	2.63×10^{-12}
YELLGB	Yellowtail flounder	4.11×10^{-10}	SKJWATL	Skipjack tuna	9.91×10^{-12}
YELLSNEMATL	Yellowtail Flounder	4.17×10^{-09}	SMOOTHOREOCR	Smooth oreo	1.95×10^{-10}
YEYEROCKPCOAST	Yelloweye rockfish	1.48×10^{-07}	SMOOTHOREOWECR	Smooth oreo	2.93×10^{-10}
			SNOSESHARATL	Atlantic sharpnose shark	3.94×10^{-13}
			SOUTHHAKECR	Southern hake	2.07×10^{-10}
			SOUTHHAKESA	Southern hake	9.87×10^{-11}
			SSTHORNHPCOAST	Shortspine thornyhead	3.32×10^{-10}
			STFLOUNNPCOAST	Starry flounder	6.20×10^{-10}
			STFLOUNSPCOAST	Starry flounder	1.31×10^{-09}
			SWORDSATL	Swordfish	4.18×10^{-11}
			TREVALLYTRE7	Trevally	4.61×10^{-10}
			VSNAPGM	Vermilion snapper	1.63×10^{-10}
			WPOLLNSO	Walleye pollock	3.29×10^{-13}
			YELL3LNO	Yellowtail Flounder	3.11×10^{-11}
			YFINATL	Yellowfin tuna	6.07×10^{-12}
			YFINCWPAC	Yellowfin tuna	2.50×10^{-12}
			YTROCKNPCOAST	Yellowtail rockfish	1.22×10^{-10}
			YTSNAPSATLC	Yellowtail snapper	4.35×10^{-10}

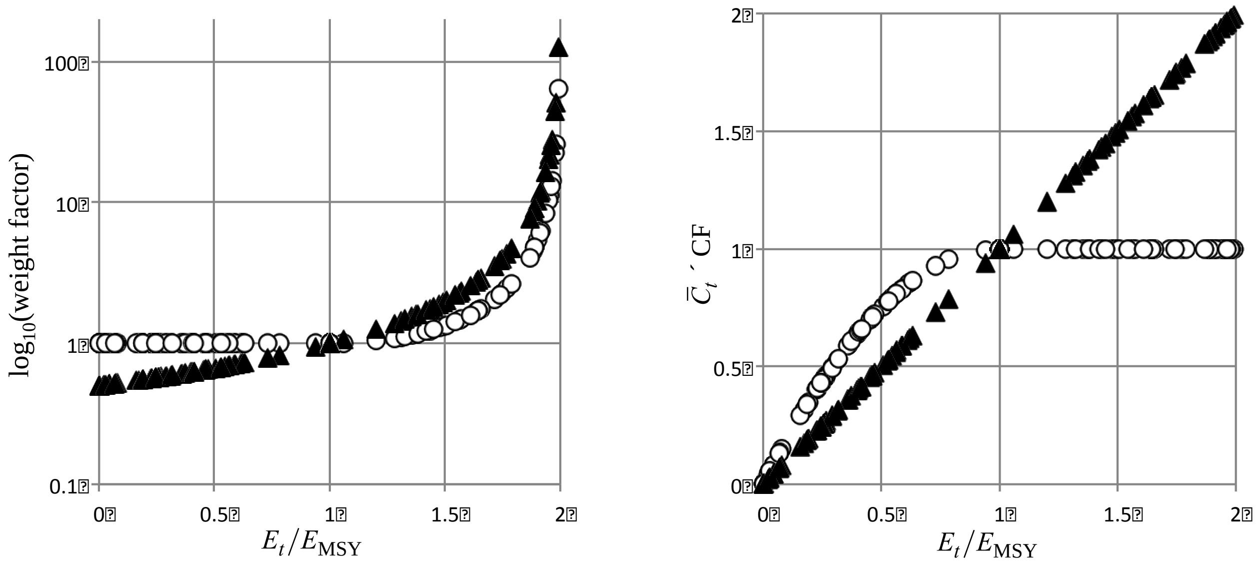

Figure 2. Weight factors (left part) and total impact (right part) according to relative fishing effort. White circle: approach proposed in Langlois et al. (2014a); black triangle: this study.

4. Discussion

This work is an improvement of the previous CF calculations in two respects:

- The CF value increases now continuously according to the total fishing effort for exploited species. The CF for a species where the current total catches is close to the MSY (i.e. close to overexploitation) is now higher than the CF for a species where catches are low, as it is shown in Figure 2, left part, when the relative fishing effort is lower than 1.
- The total impact (total catches by CF) increases continuously from low exploited species to depleted resources, whereas in the previous equations, the total impact was identical for all overexploited species (Figure 2, right part, when the relative fishing effort exceeds 1).

As already expressed in Langlois et al. (2014a), the use of MSY and steady state assumption present some limitations. The transition periods for a given stock are not properly taken into account, being it the result of a drastic increase in effort leading to overexploitation, or to a fast decrease in effort, resulting from the implementation of a low quota aimed at the rapid restoration of the stock biomass. In order to limit this inconvenient and to approximate the equilibrium state, a five-year catches average has been used.

5. Conclusion

While many impacts are now consensual in LCA and standard frameworks are starting to be available (JRC-EIS 2011), no guidelines exist for biotic impact assessment (Emanuelsson et al. 2014). This work proposes CFs at midpoint level for fishing activities, where the use of the biotic resource is taken into account in front of the resource recovery capacity (expressed by the MSY). This work allows assessment of fisheries in the LCA formalism. It contributes to a better representation of the depletion of biotic resources.

6. Acknowledgments

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Greenhouse Gas Emissions of the U.S. Diet: Aligning Nutritional Recommendations with Environmental Concerns

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ABSTRACT

Dietary choices – the overall food consumption patterns displayed by a population – can, and do, influence greenhouse gas emissions (GHGE) of the food system. Here, we consider the GHGE associated with production of the current U.S. diet, and various dietary recommendations. USDA's loss adjusted food availability dataset is used to represent current consumption. GHG emission factors were compiled from published LCA sources. Production of the current U.S. diet has GHGE of 5.0 kg CO_2eq capita^{-1}day^{-1}, 28% of which is due to food losses; a diet based on USDA's *Dietary Guidelines* has similar or greater GHGE as this current diet. Lacto-ovo vegetarian, vegan, and Harvard's Healthy Eating Plate guidelines show 33%, 53%, and 33% lower GHGE, respectively, than their USDA omnivore counterpart. Official USDA food plans at four cost levels show increasing emissions with rising food costs, but decreasing emissions per unit cost. This paper suggests aligning environmental and health objectives through dietary recommendations policy.

Keywords: sustainable diet, carbon footprint, diet shift, consumption, cost-based food plan

1. Introduction

Opportunities to reduce the greenhouse gas emissions (GHGE) associated with food and agriculture exist throughout the food production chain, but researchers warn that technological improvements in agriculture alone – enhanced yields, resource use efficiencies, etc.– will likely be insufficient to keep pace with population growth and rising demand for meat and dairy (Garnett 2011). Behavioral choices, including shifts in diet and minimizing food waste, particularly in developed countries, can have large influences. With transportation, housing and food (in that order) as the largest contributors to the carbon footprint of the typical U.S. household, often the most economically effective abatement options for consumers are dietary changes (Jones and Kammen 2011). In addition, more than 40% of food waste in industrialized countries happens at the retail and consumer levels (Gustavsson et al. 2011). In this study, we explore in detail the GHGE associated with current and recommended diets in the U.S.

Changes in both the quantity and quality of the American diet may hold potential to affect the carbon footprint of food production. The current obesity epidemic, in the U.S. as well as other developed nations, garners broad academic, political and media attention; the most recent statistics claim that 69% of U.S. adults are overweight and 36% are obese (CDC 2013). Food overconsumption and obesity contribute not only to human health dangers, but also translate directly and indirectly into increased agricultural demand, excess resource utilization, and concomitant environmental impacts (Blair and Sobal 2006; Cafaro et al. 2006). Not only are Americans over-consuming, they are not eating properly: repeated assessments find that Americans do not meet the Federal dietary recommendations (Krebs-Smith et al. 2010; Wells and Buzby 2008). Dietary choices influence the environmental cost of food consumption (Carlsson-Kanyama and Gonzalez 2009; Marlow et al. 2009), and research from Germany (Meier and Christen 2013) and Austria (Fazeni and Steinmuller 2011) suggests that a shift from current consumption patterns to German Nutrition Society dietary recommendations can result in reduced environmental impacts from the agri-food sector. Dietary choices are complex and influenced by a myriad of factors; dietary recommendations presented by governments are but one avenue to influence policy and consumer choices to move toward a more nutritionally healthy diet. However, if multiple objectives (e.g., health and sustainability) are not aligned in recommendations, undesirable outcomes are possible. Here, we explore the implications on food related GHGE of U.S. dietary recommendations and policy-related food patterns.

2. Methods

2.1. U.S. food consumption and losses

The Loss Adjusted Food Availability (LAFA) data series (USDA ERS 2012) serves as a useful proxy for per capita food consumption in the U.S. The food availability series measures the use of basic food commodities (e.g., wheat, beef, fruit, vegetables) by tracking their "disappearance" in the U.S. marketplace. For most commodities, the available supply is the sum of production, imports and beginning stocks, minus non-food use (feed and seed, industrial uses), exports, and ending stocks for a given calendar year. In the Loss Adjusted data series, the food availability data for over 200 commodities are modified by percent loss assumptions at the primary level, retail/institution level, and consumer level. Retail losses include dented cans, unpurchased holiday foods, spoilage, and the culling of blemished or misshaped foods. Consumer losses include spoilage, cooking shrinkage, and plate waste. Nonedible food losses (bones, peels, pits, etc.) are accounted for separately in the data series and not included in the results presented here. We use LAFA data from 2010 in this study to represent current food consumption patterns in the U.S., shown in Table 1. Percent losses for individual foods were assumed to be unchanged in considering a shift to dietary recommendations, thus allowing an assessment of how food losses might be affected by a dietary shift. Further detail and report of per capita availability and loss estimates for various foods are presented in (Heller and Keoleian 2014).

2.2. Dietary recommendations and food plans

The 2010 *Dietary Guidelines for Americans* contains food pattern recommendations (Appendix 7, 8 and 9 of (USDA 2010)) that provide recommended average daily intakes of different food groups (assumed to be in nutrient dense forms and lean or low-fat) for a range of Calorie levels, depending on individual caloric intake needs. The *Dietary Guidelines* also provide estimated caloric needs by age, gender and physical activity level (Appendix 6 of (USDA 2010)). Weighting these estimated caloric needs with U.S. age and gender demographics from 2010 (U.S. Census Bureau 2011) suggests that the weighted average caloric need for a moderately active U.S. population is 2093 Calories[1] (1867 Calories and 2361 Calories for sedentary and active, respectively). For the purpose of this paper, we explore recommended food patterns at two caloric levels (see Table 1): the current caloric intake (according to the LAFA series) of 2534 Calories, representing an iso-caloric diet shift, and the often used reference point of 2000 Calories, representing a weighted average target intake for the average U.S. citizen, assuming moderate activity. We also evaluate lacto-ovo vegetarian and vegan food pattern recommendations at 2000 Calories, as defined by the *Dietary Guidelines* (USDA 2010).

The Healthy Eating Plate (HEP) diet is a food pattern designed to represent the Alternative Healthy Eating Index (AHEI) developed by nutrition experts at the Harvard School of Public Health. AHEI is based on foods and nutrients predictive of chronic disease risk, and has demonstrated a stronger association with chronic disease risk, particularly coronary heart disease and diabetes, than USDA's Healthy Eating Index (which quantifies adherence to *Dietary Guidelines*) in prospective cohort studies (Chiuve et al. 2012). The HEP is characterized by high quality grains (whole vs. refined), healthy proteins (fish, poultry, beans and nuts vs. red meat and processed meat), greater intake of poly unsaturated fatty acids (healthy oils), reduced intake of sugar-sweetened beverages, and reduced dairy (Harvard School of Public Health 2014). The HEP food pattern at 2000 Calories examined here is defined in Table 1 and is from (Willett 2014).

USDA also maintains official food plans designed to meet required dietary standards and specified cost levels, which are used for various policy implementation purposes. The Thrifty Food Plan (Carlson et al. 2007b) serves as a national standard for a nutritious diet at a minimal cost and is used as the basis for maximum food assistance allotments. Bankruptcy courts often use the value of the Low-Cost Food Plan (Carlson et al. 2007a) to determine the portion of a bankrupt person's income to allocate to necessary food expenses; the U.S. Department of Defense uses the value of the Liberal Food Plan to determine the Basic Allowance for Subsistence rates for all service members; many divorce courts use the values of the USDA Food Plans to set alimony payments, and they are also used to set child support guidelines and foster care payments.

These food plans reflect the consumption patterns and eating habits of U.S. citizens as determined by the

[1] 1 Calorie = 1 food calorie = 1 kilogram calorie = 4,186.8 Joules

2001-2002 National Health and Nutrition Examination Survey (NHANES); the U.S. population is divided into quartiles of food spending by combining NHANES food intake data with the 2001-2002 Food Price Database (Carlson et al. 2007a). The Thrifty Food Plan corresponds with the first quartile of food expenditure, the Low-Cost plan with the second quartile, the Moderate-Cost Plan with the third quartile, and the Liberal Food Plan with the upper quartile. A mathematical optimization model is used to identify food market baskets representing a diet as close as possible to actual food consumption patterns within a given quartile group while also meeting USDA dietary standards and the cost limits for that quartile. Within each Food Plan, diets for 15 age-gender groups are identified; for simplicity's purposes, we consider here only the male and female diets in the 19-50 year old group. Purchasing costs at the four quartile levels are updated monthly (USDA 2014); costs from June 2013 (assigned by USDA to represent the annual average) are used here in order to present GHGE per dollar for the four food plans. Note that updated Food Plans reflecting the (most recent) 2010 Dietary Guidelines have not yet been published; the Food Plans reported in (Carlson et al. 2007a; Carlson et al. 2007b) are based on the 2005 Dietary Guidelines for Americans.

Table 1. Per capita daily intake definitions for the current US diet and recommended food patterns.

	Food pattern equiva-lents units	Cur-rent US diet[a]	Food Pattern Recommendations from 2010 Dietary Guidelines (USDA)[c]				Healthy Eating Plate food pattern[j]	GHGE of repre-sentative food(s)[l]
			omnivore		Lacto-ovo vegetarian	vegan		
		Per capita per day						kg CO_2eq. (food pattern unit)$^{-1}$
Total Calories	Cal.	2534	2534[e]	2000	2000	2000	2000	
Fruits	Liter eq.	0.19	0.47	0.47	0.47	0.47	0.47	0.76
Vegetables	Liter eq.	0.38	0.78	0.59	0.59	0.59	0.59	-
dark green veg	Liter eq.	0.050	0.071[d]	0.047[d]	0.047[d]	0.047[d]	0.085[d]	0.12
red & orange veg	Liter eq.	0.059	0.24[d]	0.19[d]	0.19[d]	0.19[d]	0.20[d]	0.40
beans & peas	Liter eq.	0.028	0.071[d]	0.047[d]	0.047[d]	0.047[d]	0.069[d]	0.25
starchy veg	Liter eq.	0.15	0.24[d]	0.17[d]	0.17[d]	0.17[d]	0.069[d]	0.14
other veg	Liter eq.	0.10	0.19[d]	0.14[d]	0.14[d]	0.14[d]	0.17[d]	0.40
Grains	kg-eq.	0.21	0.25	0.17	0.17	0.17	0.20	0.49
Protein foods	kg-eq.	0.21	0.18	0.16	0.16	0.16	0.18	-
seafood	kg-eq.	0.012	0.040[d]	0.031[d]	0	0	0.031[d]	6.4
meat, poultry, eggs	kg-eq.	0.17	0.12[d]	0.10[d]	0.016[d]	0	0.077[d]	-
beef	kg-eq.	0.054	0.040[f]	0.031[f]	0	0	0.0082[d]	26
pork	kg-eq.	0.037	0.026[f]	0.023[f]	0	0	0.0082[d]	6.7
poultry	kg-eq.	0.068	0.048[f]	0.040[f]	0	0	0.048[d]	4.9
eggs	kg-eq.	0.014	0.011[f]	0.0085[f]	0.016[d]	0	0.012[d]	6.3
nuts & seeds	kg-eq.	0.024[b]	0.020[d,g]	0.017[d,g]	0.054[d]	0.059[d]	0.043[d]	0.85
soy products	kg-eq.	-[b]	-[g]	-[g]	0.048[d]	0.040[d]	0.031[d]	3.5
beans & peas	kg-eq.	-[i]	-[i]	-[i]	0.040[d, i]	0.054[d, i]	-[i]	2.1
Dairy	Liter eq.	0.35	0.71	0.71	0.71	0.71[k]	0.24	1.69 (0.85[k])
Oils	g	46.2	33.0	27	19	19	50	0.0063
Max. SoFAS[h]	Cal.	513	351.4	258	258	258	120	0.00035

[a]from: (USDA ERS 2012) http://www.ers.usda.gov/datafiles/Food_Availabily_Per_Capita_Data_System/LossAdjusted_Food_Availability/servings.xls; reported in cup eq. and oz. eq.
[b]source dataset only gives nuts and does not explicitly differentiate seeds or soy products.
[c] Appendix 7,8 & 9 (USDA 2010).; reported in Cup eq. and oz. eq.
[d] presented in guidelines as weekly recommendations. Averaged here to daily intakes.
[e] Food pattern for 2534 Calories derived by linear interpolation between 2400 and 2600 Calorie pattern from (USDA 2010).
[f] *Dietary Guidelines* only report the aggregated "meat, poultry & eggs" category. Subcategory distributions presented here are based on current consumption patterns.
[g] Subcategories are combined in *Dietary Guidelines* (as "nuts, seeds & soy products"), and this combined category is presented here under "nuts & seeds."
[h] max. SoFAS = maximum solid fats and added sugars, but is composed only of added sugars in the Healthy Eating Plate guidelines (HEP does not set a maximum limit of calories from healthy fats).
[i] this "beans & peas" category represents *additional* intake for vegetarian and vegan diets, on top of the beans & peas recommended as part of vegetable consumption.
[j] Healthy Eating Plate food pattern from Walter Willett, personal communication (Willett 2014).
[k] The vegan "dairy group" is composed of calcium-fortified beverages and foods from plant sources. Soy milk is used as a GHGE proxy for the entire group.
[l] GHGE factors for food groups were calculated as described in (Heller and Keoleian 2014), Supporting Information.

2.3. Food GHGE data

While numerous LCA-based studies of food production have been published, there currently is not a comprehensive and authoritative database of food environmental impact. Thus, we have chosen a meta-analysis approach of published LCA data to arrive at representative GHGE factors for the diversity of foods considered in this study. Results are drawn from a variety of sources (referenced in (Heller and Keoleian 2014), Supporting Information (SI)), compiled by food type, and averaged across comparable food type. These sources include studies representing a variety of countries of origin, climatic conditions, transportation distances, and production methods and therefore are intended to provide a reasonable range of expected values rather than a definitive result for each food type. U.S. based data is limited, and thus this meta-analysis includes data from other developed countries. The average GHGE values for the ~100 foods considered in this study, along with minimum and maximum GHGE values from the meta-analysis, are given in (Heller and Keoleian 2014), SI. In some cases where insufficient data on individual foods exist and where small differences in GHGE are expected between foods (such as, e.g., added sugars and sweeteners), available examples are used to represent the entire food category.

The emission factors for broad food categories and subcategories shown in Table 1 are averages of representative foods, weighted by current consumption patterns in the U.S. as dictated by the LAFA dataset. Conversions to food pattern equivalents units are based on grams per food pattern equivalents contained in the LAFA dataset (USDA ERS 2012).

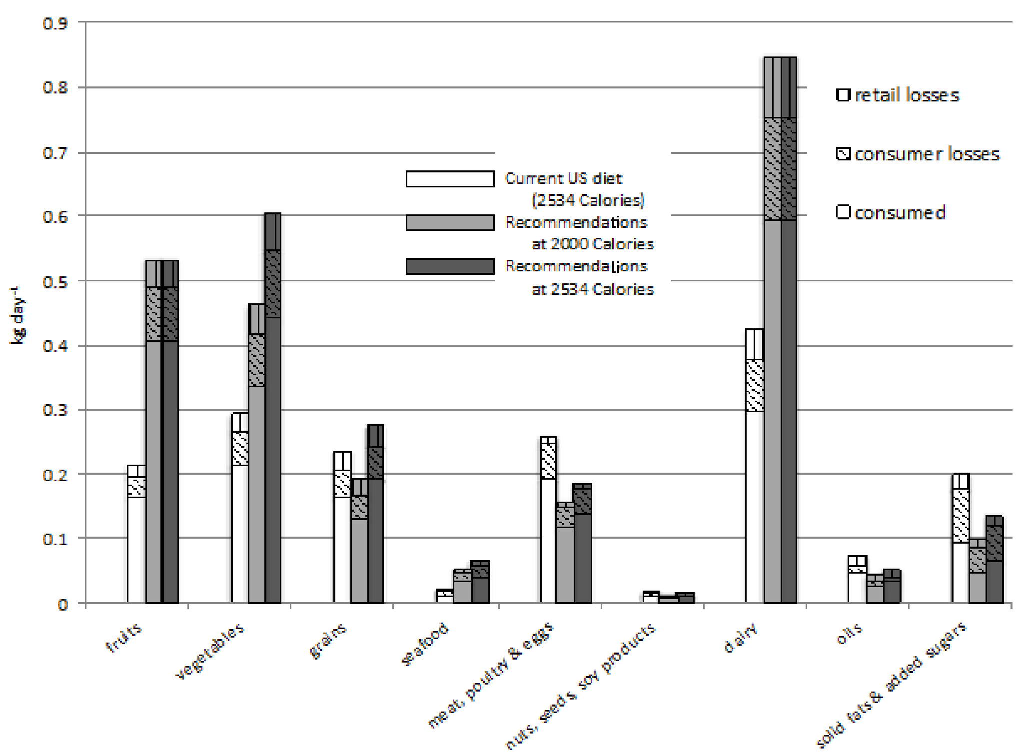


Figure 1. Per capita quantities consumed and losses by food types for the current US diet (white bars), a recommended diet at 2000 Calories (light grey bars) and a recommended diet at 2534 Calories (dark grey bars).

3. Results

Figure 1 summarizes the average daily per capita diet in the U.S. as well as food patterns corresponding with the USDA dietary recommendations. To shift their diet to meet recommendations, the average U.S. citizen would need to significantly increase fruit, vegetable and dairy intake while decreasing intakes of meats, oils and solid fats and added sugars. Figure 1 also shows the retail- and consumer-level food losses, as specified by the LAFA dataset, and expected food losses with a recommended diet, assuming the same loss percentages for individual food types. The total weights of food consumed and wasted are reported in Table 2. In Figure 2, the same diets are shown but with individual food weights multiplied by GHGE factors, providing a distribution of GHGE associated with producing the U.S. diet. Not surprisingly, meats and dairy dominate the carbon footprints of both current and recommended diets. As can be seen in Table 2, the total diet-related GHGE is greater for dietary recommendations at the same caloric intake compared with the current diet, whereas a shift to recommendations accompanied by a 20% decrease in caloric intake shows the same GHGE as the current diet. This appears to be primarily due to the balance between decreased meat and increased dairy in the recommended diets. The total weight of food losses associated with a recommended diet at 2534 and 2000 Calories is nearly 50% and 30% greater than the current diet, respectively, whereas GHGE associated with those food losses only increase 20% and 7%, respectively. The weight of food losses increases primarily due to increased consumption of fruits, vegetables and dairy – all foods with relatively high loss percentages but low GHGE factors; decreases in meat consumption also contribute to the disproportional increase in food loss GHGE.

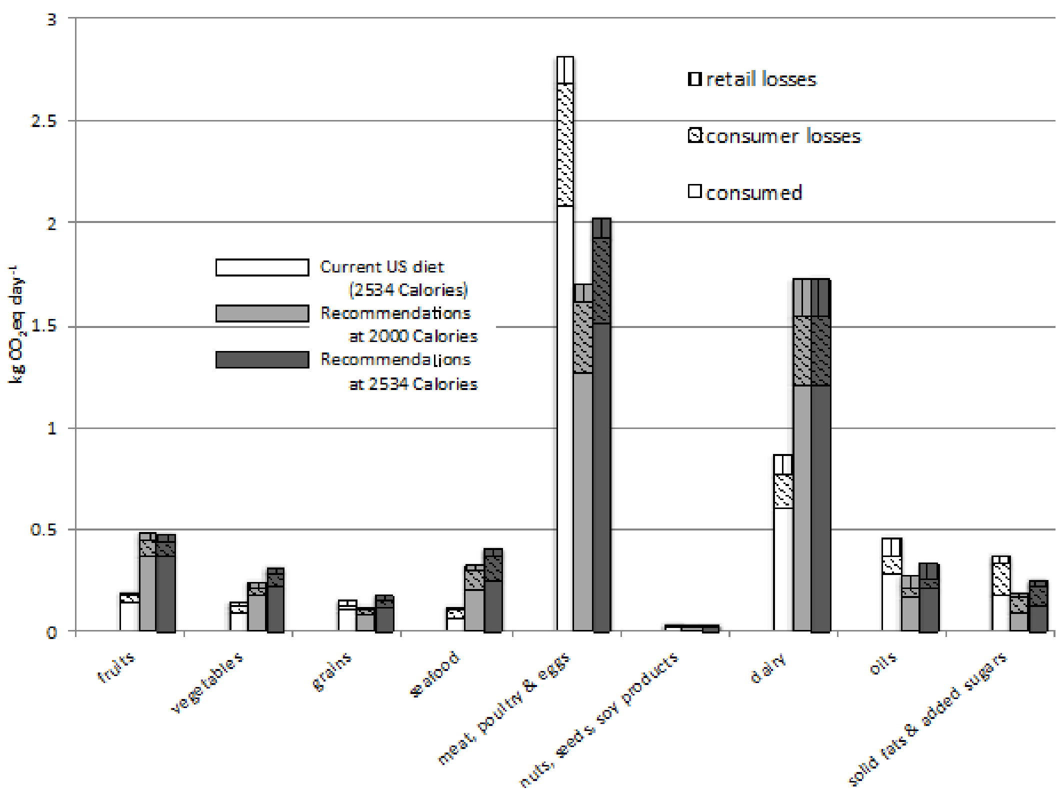


Figure 2. Per capita greenhouse gas emissions associated with producing consumed food and losses by food types for the current US diet (white bars), a recommended diet at 2000 Calories (light grey bars) and a recommended diet at 2534 Calories (dark grey bars).

Table 2. Total per capita weights and GHGE associated with consumed food and estimated food losses of the diets shown in Figures 1 and 2.

	weight (kg day^{-1})				GHGE (kg CO_2 eq day^{-1})			
	consumed	Retail losses	Consumer losses	Total losses	consumed	Retail losses	Consumer losses	Total losses
Current US diet (2534 Cal.)	1.19	0.17	0.36	0.53	3.60	0.40	1.0	1.4
US dietary recommendations at 2000 Cal.	1.70	0.24	0.46	0.69	3.57	0.45	1.1	1.5
US dietary recommendations at 2534 Cal.	1.91	0.27	0.53	0.79	4.02	0.51	1.2	1.7

Figure 3 compares the GHGE associated with food patterns for four recommended diets all at 2000 Calories: the USDA recommended diet shown in Figures 1 & 2, lacto-ovo vegetarian and vegan adaptations of this diet (also offered by USDA), and a food pattern representing the HEP recommendations made by Harvard School of Public Health. As expected, limiting or eliminating animal products in the vegetarian and vegan diet significantly reduces GHGE. Interestingly, the HEP diet, which recommends reduced (but not eliminated) levels of animal proteins and dairy, demonstrates GHGE equivalent to the lacto-ovo vegetarian diet.

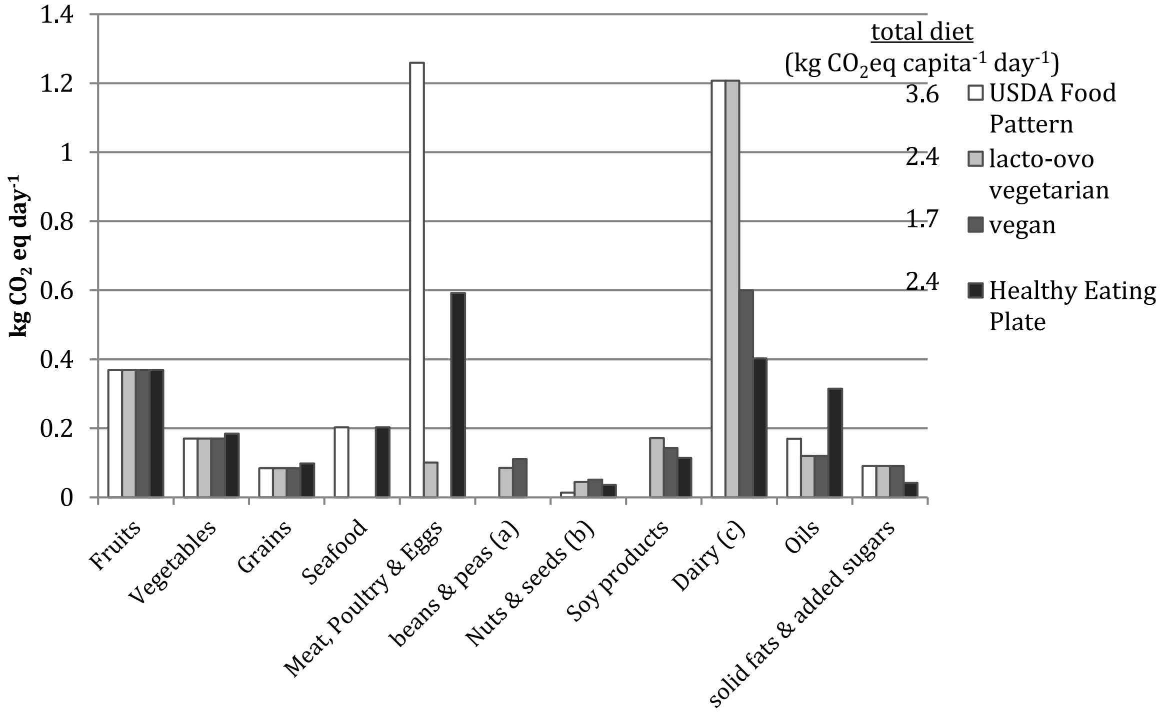
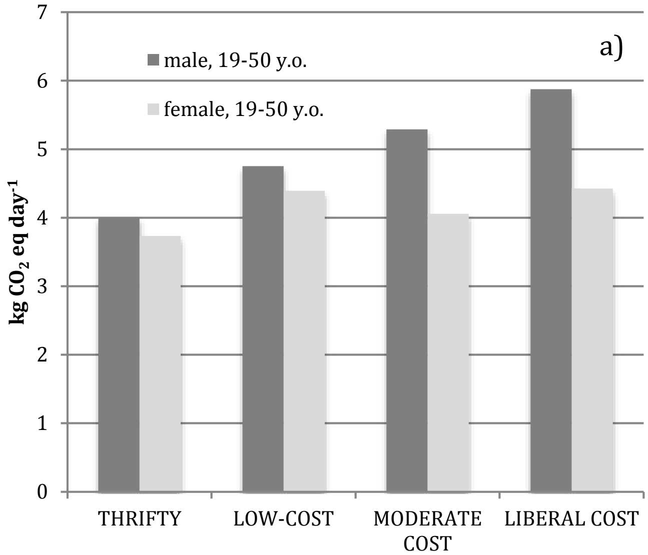


Figure 3. Comparison of the per capita greenhouse gas emissions associated with various 2000 Calorie recommended food patterns. Diets are as described in Table 1. Notes: (a) this category represents beans and peas added to the vegetarian and vegan diet *in addition* to those included in "vegetables". (b) nuts, seeds & soy products are combined into the same category in the USDA food pattern, presented here as a bar only in "nuts & seeds". (c) "dairy" category in the vegan diet contains calcium-fortified beverages and foods from plant sources.

Figure 4 offers a high-level summary of the GHGE associated with USDA cost-based food plans. These food plans are based on the food expenditures of surveyed U.S. citizens. Food expenditure distributions for each of 15 age and gender categories are divided into quartiles; the average share of food consumption in each quartile is used to represent the group. An optimization is then performed to arrive at a food plan (collection of food items) that meets nutritional requirements, does not exceed the expenditure cut-off for the group, and minimizes change in consumption from the average food consumption for the group. Figure 4 shows that the food-plan-associated GHGE for 19-50 year old men and women rise with food expenditures, but apparently at a slower rate than the increasing cost, as the GHGE per $ cost decreases. Table 3 gives the distribution of both food expenditures and GHGE across broad food categories for all four food plans. Interestingly, the dominant cause of the increase in GHGE from the Thrifty to the Low-cost food plan is due to the introduction of soft drinks and sodas: these beverages are absent from the Thrifty food plan for 19-50 year olds. Additional trends are more difficult to identify: increasing red meat consumption with increased cost plays a role, often buffered by decreasing poultry consumption. Vegetable consumption tends to move away from starchy vegetables like potatoes to dark green and other vegetables with increasing costs. Surprisingly, legumes (beans, lentils, peas), which might be considered low cost protein sources, actually increase with expenditure in these food plans.



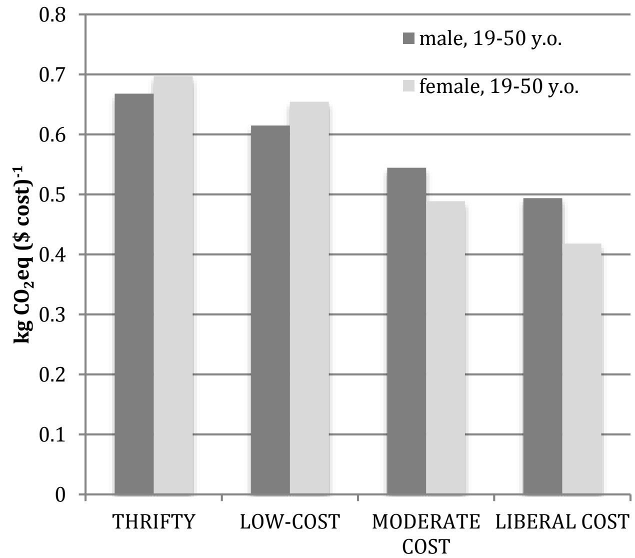


Figure 4. Comparison of GHGE associated with USDA food plans developed at meal cost quartiles, shown for diets developed for 19-50 year old males and females.

Table 3. Expenditure (cost) and GHGE shares across food categories for Thrifty, Low-cost, Moderate-cost and Liberal cost food plans. Food plans for 19-50 year old males and females are shown. Expenditure shares from (Carlson et al. 2007a; Carlson et al. 2007b).

	THRIFTY		**LOW**		**MODERATE**		**LIBERAL**	
	M	**F**	**M**	**F**	**M**	**F**	**M**	**F**
% COST								
grains	14.4	16.6	15.1	12.2	21.4	18.3	19.8	16.4
vegetables	27.1	21.9	23.9	23.3	24.0	30.8	28.1	27.1
fruits	19.8	17.9	17.9	20.6	14.9	14.6	13.4	18.4
milk products	14.1	16.4	13.5	15.3	12.4	14.1	11.5	14.3
Meat[a]	16.5	20.8	20.5	21.6	21.8	19.0	21.4	19.3
other foods[b]	8.1	6.4	9.0	7.2	5.6	3.3	5.8	4.4
% GHGE								
grains	4.7	3.1	2.9	2.7	3.7	3.5	3.7	3.5
vegetables	8.3	6.6	7.5	6.4	7.4	10.0	8.4	10.1
fruits	9.4	6.3	7.4	8.1	7.3	7.4	7.2	10.9
milk products	26.7	28.4	24.8	27.5	23.9	31.1	24.3	30.6
Meat[a]	44.4	50.6	39.3	39.3	47.5	42.4	45.3	36.7
other foods[b]	6.4	5.0	18.1	16.0	10.2	5.5	11.1	8.3

[a]"Meat" include beef, pork, veal, lamb, poultry, fish, sausages and luncheon meats, nuts, nut butters and seeds, and eggs and egg mixtures.
[b]"Other foods" include table fats and oils, gravies, sauces and condiments, coffee and tea, soft drinks and sodas, sugars sweets and candies, ready-to-serve, condensed and dry soups, and frozen or refrigerated meals.

4. Discussion

Better understanding the environmental implications of dietary food patterns is a critical step toward a more sustainable food system. Here, we provide an approach to estimating the GHGE associated with the current U.S. diet. The LAFA dataset on which our estimate is based is a top down approach, starting with *disappearance* of (primarily) agricultural commodities and, through a series of loss estimates, arriving at a proxy for per capita consumption. Some impacts of food processing and distribution are missed in this approach, and we acknowledge that the GHGE factors used for individual foods are limited in their specificity and geographic accuracy. While our estimate for GHGE associated with the average U.S. diet of 5.0 kg CO_2eq capita^{-1} day^{-1} (consumption plus losses) is within the range of other national average diet carbon footprints reported in the literature (see Table 3 in (Heller et al. 2013)), reported economic input-output LCA based estimates for the U.S. diet are higher (8.4 kg CO_2eq. capita^{-1}day^{-1} from (Weber and Matthews 2008) and 8.5 kg CO_2eq. capita^{-1}day^{-1} from (Jones and Kammen 2011)).

Despite the limitations, the approach presented here is valuable in comparing the relative effects of dietary shifts. A USDA recommended dietary pattern at the same caloric intake as the current diet is associated with 12% greater GHGE. While a 20% reduction in caloric intake at a constant dietary pattern would result in a 20% decrease in GHGE, a 20% reduction in caloric intake combined with a shift to dietary recommendations results in no change in GHGE.

Food losses represent a significant impact for the U.S. food system. By our estimates, the GHGE associated with retail- and consumer-level losses totaled 160 MMT CO_2 eq. in 2010, roughly equivalent to the emissions of 33 million average passenger vehicles (Heller and Keoleian 2014). Interestingly, this is also roughly equivalent to the GHGE difference between the omnivore recommended USDA food pattern and the lacto-ovo vegetarian food pattern (Figure 3). A population-wide shift to vegetarianism is highly unlikely in the U.S., but comparisons with the HEP guidelines suggest that similar reductions are possible without completely giving up meat. Reduced dairy consumption is a primary driver of the carbon footprint differences seen with the HEP food pattern. While the scientific debate over the health implications of dairy in our diet (e.g., (Ludwig and Willett 2013)) are beyond the scope of this paper, clearly the trade-offs between the health and environmental impacts of dairy deserves further attention, and is a focus of our future work. Of course, in order to infer actual change in climate impact from the dietary shifts considered in this paper, a consequential LCA analysis is needed, as market-mediated and other indirect effects are likely significant. Still, the attributional approach presented here suggests the potential for reduction in food system GHGE through changes in diet.

Analysis of the USDA cost-based food plans offers an initial glimpse at consideration of a triple bottom line objective for dietary recommendations. It should be noted that these food plans are based on a different approach to defining current U.S. consumption patterns than the diet presented in Figures 1 and 2, and therefore may not be directly comparable. The influence of current consumption patterns on these food plans should also not be underestimated: they do not represent an optimal low-cost nutritious diet, but instead a nutritious diet within a given cost constraint that *minimally deviates from current consumption*. That said, the results in Figure 4 further demonstrate the effect that dietary composition can have on GHGE of food plans that are (roughly) nutritionally equivalent. In addition, it raises questions as to what the ideal optimization objective might be when including environmental concerns in nutritional policy and recommendations: minimizing overall environmental impact? Impact per unit cost? Impact per unit nutritional benefit? Such questions echo the perennial challenge of functional unit definition in food LCA, and are at the heart of the additional work needed to permit meaningful incorporation of environmental indicators into nutritional policy.

5. Conclusion

The GHGE associated with production of the food consumed by the U.S. population is significant: 8% of total U.S. GHGE in 2010, based on our estimates (Heller and Keoleian 2014). Yet, the results presented here suggest that these emissions could be reduced by upwards of 30%, not by eliminating all animal products from the diet, but by shifting to a healthy diet based on foods and nutrients that are predictive of lowered chronic disease risk. According to our analysis, a diet based on the food patterns recommended by the *Dietary Guidelines for Americans 2010* would have the same or greater GHGE as the current U.S. diet. Absent of additional interventions, greater food losses can be expected with a shift to dietary recommendations, but the relative GHGE impact of those losses are diminished. While policy-level food plans designed to supply ample nutrition at limited cost show increasing GHGE with increasing cost, they also demonstrate the highest emissions per unit of food cost at the lowest expenditure food plan, suggesting that further optimization of the health/ environment/ affordability triple bottom line is possible. Policy-level discourse regarding sustainable food consumption and the inclusion of environmental factors in nutritional guidelines has emerged in various European countries as well as in Australia (Lang and Barling 2013), but such conversation is at a nascent stage in the U.S. This study provides much needed quantitative evidence to the emerging U.S. sustainable diet debate, and suggests the need to align environmental and health objectives of the U.S. food system through dietary recommendations.

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Using LCA to Identify Options for Greenhouse Gas Emission Reductions in Australian Wheat Farming

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ABSTRACT

Agriculture can be a contributor to a nation's total greenhouse gas emissions. Nitrapyrin (2-chloro-6-trichloromethylpyridine) is a nitrification inhibitor used globally to improve the efficacy of applied ammoniacal nitrogen fertilizers. By stabilizing nitrogen in the ammoniacal form, it results in more of the applied nitrogen available to plants thereby reducingN_2O and nitrate emissions from the farm. We used LCA to determine if the potential benefit from the use of nitrapyrin exceeded the burdens to create and transport it, and to discover additional opportunities to reduce emissions from high rainfall zone wheat farming in Australia. Use of nitrapyrin could reduce global warming potential (GWP) by 20% and marine eutrophication by 16%, with burdens created by its supply of only 0.5% and 0.01%, respectively. Production of nitric acid for urea ammonium nitrate fertilizer impacts GWP as much or more than the field emissions. Reducing emissions from nitric acid production may provide a similar (15%), independent and additive, benefit in GWP.

Keywords: nitrapyrin, greenhouse gas emissions, nitric acid, wheat, high rainfall zone

1. Introduction

Climate change, driven by emissions of "greenhouse gases" (GHG) such as CO_2, CH_4 and N_2O, is a critical global issue, but was particularly of concern in Australia. About 93% of electricity in Australia is generated from fossil resources, primarily coal (International Energy Agency, 2012), which generates greenhouse gas emissions and is a challenge to change over the short term. There was great interest and research on various approaches to help Australia meet targets for GHG reduction set out by the Kyoto protocol; one of them was to increase use of nitrification inhibitors to reduce nitrous oxide (N_2O) emissions from agriculture (Dalal *et al*., 2003; Bhatia *et al*., 2010).

Nitrapyrin (2-chloro-6-trichloromethylpyridine) is a nitrification inhibitor produced in Pittsburg, California, by Dow AgroSciences, a business unit of The Dow Chemical Company (Dow), and is used globally in agriculture as a way to improve the efficiency and efficacy of applied fertilizers (Dow, 2012). Nitrification is a natural biological process that converts ammonia (from fertilizers) in soils to nitrate. The nitrate form of nitrogen is highly susceptible to losses from the soil root zone as N_2O emissions to air and nitrate emissions to ground and surface waters. The pathways of emissions are shown in Figure 1; nitrapyrin inhibits the nitrification process and thus reduces these emissions (Wolt, 2000; Chen *et al*., 2010). Reduction in N_2O emissions from fertilizer by use of nitrapyrin has been demonstrated in many studies (Wolt, 2004; Akiyama, Yan and Yagi, 2010).

The goal of our work was to determine if the potential benefit from the use of nitrapyrin exceeded the burdens to create and transport it, to understand other potential impacts, and to discover additional opportunities to reduce GHG emissions from wheat farming. Life Cycle Assessment (LCA) is a useful methodology for examining the total environmental impact of a product or service. LCA takes a holistic view, examining environmental impacts over the complete "cradle to grave" product life cycle. Results from LCA address the complete environmental impacts of a product, and are hence more meaningful than those obtained for a single process or step in the life cycle. A life cycle perspective helps to ensure that environmental burdens are not unintentionally transferred from one life cycle stage to another and helps to prevent unintended environmental consequences.

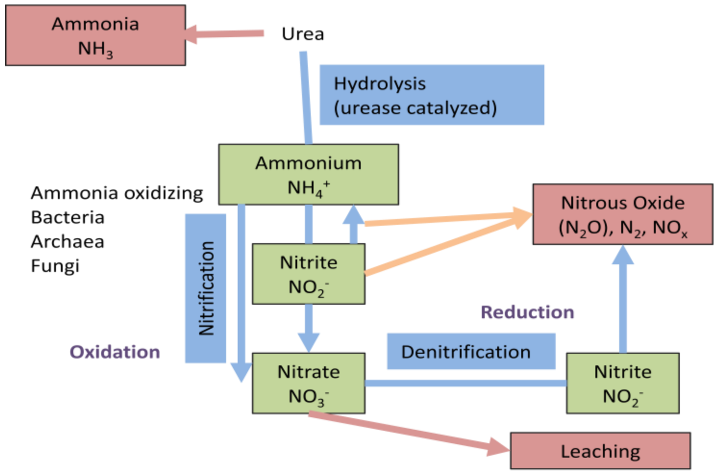

Figure 1. Routes to key emissions from use of fertilizers (re-drawn from a 2012 presentation by Chen and Suter)

Two studies were particularly useful in understanding the inputs to and emissions from wheat farming in Australia. Chen et al. (2010) reports the impact of use of nitrification chemicals on nitrogen speciation and emissions from laboratory experiments. It provided a basis for the unit ratio of nitrapyrin to applied N, applied N rate, and for the impact of nitrapyrin on N_2O emissions. Biswas *et al*. (2010) had extensive data and analysis of inputs and emissions from HRZ (high rainfall zone) wheat in rotation farming. Although there was no nitrogen fertilizer used due to the rotation with sheep farming, the other inputs were quite relevant to this project. The report also included field data for N_2O emissions. HRZ farming is distinct from wheat farming in, for example, Western Australia, which has been the subject of previous publications related to LCAs (Biswas *et al*, 2008; Biswas *et al*., 2010).

2. Methods

LCA is framed by ISO standards 14040 and 14044 (ISO, 2006a; ISO, 2006b), which provide comprehensive guidelines for conducting an LCA study. Good insights on the uses and limitations of LCA are described in many publications (EU-JRC, 2010; Curran, 2006; Curran, 2012). Previous use of LCA at Dow is also described in publications (DiMuro *et al*., 2014; Helling and Russell, 2006; Helling *et al*, 2012).

SimaPro 7.3.3 from Pré Consultants was the life cycle assessment software used in this study. Dow and other data were used directly to create process models in SimaPro. The primary sources of information for nitrapyrin production were Dow technical experts and Dow manufacturing databases. Dow confidential statements of formulation were used for the composition of the delivered product (also referred to by the tradename "eN-trench™"). Ecoinvent v2.2, was used within SimaPro to model utility process operations, transportation, packaging, and other material inputs. For materials produced or operations conducted in Australia, such as fertilizers, a version of the Ecoinvent data based adapted for Australian conditions (primarily for the sources of heat and power) was used (v 2012.5) (Life Cycle Strategies, 2012). Life cycle impact assessment (LCIA) was performed using valuation systems available in Ecoinvent, primarily selected from ReCiPe (Hischier *et al*., 2009; Goedkoop *et al*., 2010), using "midpoint" metrics and with no normalization to a target or weighting of different impact categories. The reported results include the impact categories of global warming potential (GWP), cumulative energy demand, water withdrawals, acidification potential (AP), freshwater and marine eutrophication potential (EP), photochemical oxidant creation potential (POCP) (smog), and ozone depletion potential (ODP).

The function of nitrapyrin is to reduce losses of fertilizer nitrogen by nitrification pathways, thereby preventing losses of the nitrogen fertilizer in the crop. Nitrapyrin is marketed and sold on the basis of a treated hectare, so the functional unit was one hectare of high rainfall zone (HRZ) wheat farming in Australia, with and without applied nitrapyrin. Calculation of potential impacts per ton of wheat production was done as part of the sensitivity analysis.

This was a cradle-to-grave study, so the boundaries extended upstream to materials in the earth and continues to materials returned to the earth (air, water, or soil). The primary chemical reaction in the process is the chlorination of alpha-picoline to make 6-chloro-2-trichoromethylpyrine ("α-6-tet"), the active ingredient in nitrapyrin. A high-level view of the overall system is shown in Figure 2. The figure shows the flow of the primary mass of the product; there are mass and energy inputs to each of these blocks and potentially emissions and by-products from each block. In the figure, TG stands for "technical grade" and AU is the country code for Australia. Capital equipment was excluded from the study, except for farm equipment, for which data were available in Biswas *et al.* (2010). Transportation was included for all significant mass flows or any transport over long distances. TG production, formulation, and packaging are all done in multi-product plants, for which the consumption of utilities and emissions (which are tracked in Dow per facility) were allocated equally by mass to all the production. Raw material consumption was known in detail specifically for each material; no material inputs were excluded, although minor inputs totaling <1% of the formulated products were lumped together as "chemicals, organic".

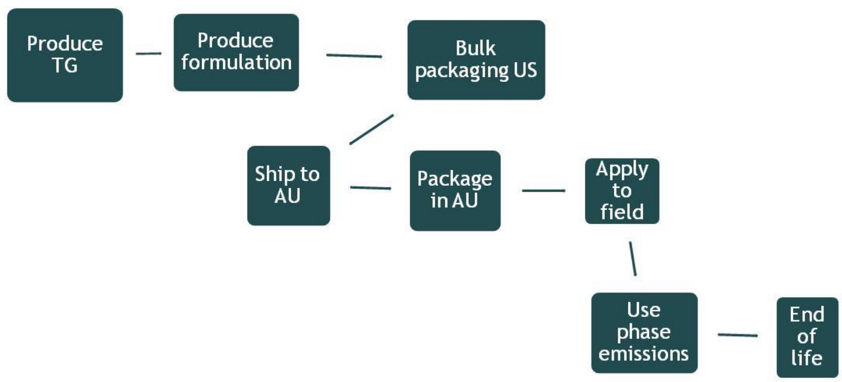

Figure 2. High-level process map for supply and use of nitrapyrin

The data sources and assumptions related to the direct emissions from applied chemicals were important for this project. Direct fossil CO_2 emissions from use of farm chemicals came from two sources: Urea and nitrapyrin, with urea being more significant by a factor of about 50. We assumed that all of the CO in the applied urea was converted to CO_2, as per IPCC guidelines, and also that all of the carbon in the applied nitrapyrin was released as CO_2 (this is expected to be a conservative estimate). N_2O emissions for the no-nitrapyrin case were taken from IPCC (2006) models applied to wheat, with the key inputs (besides the urea-ammonium nitrate (UAN) molecular weight, elemental composition, and application rate) being:

- Nitrogen fraction of crop residues, 0.0064
- Crop dry matter fraction, 0.85
- Mass ratio of residues to crop, 2.3, which was assumed to all be left on the field

The resulting N_2O emissions from fertilizer use and crop residues are 3.95 kg/ha. N_2O emissions were also calculated from the other nitrogen containing components in the formulation, and were less than 0.1% of those from the urea.

There is a range in the reported benefit of using nitrapyrin, as it depends on soil moisture, temperature and other factors (Dalal, *et al*., 2003). Wolt (2004) found an average reduction in field GHG emissions (N_2O and CH_4) of 51% in fourteen studies covering a range of crops, locations and applied fertilizer type; Chen *et al*. (2010) found a reduction in N_2O emissions of 65-98% in laboratory work covering a wide range of soil temperature and moisture levels. We believe Chen's work to be of high quality, more recent, and particularly relevant for Australia. To be conservative, we selected the 65% minimum reduction in N_2O emissions from this work.

3. Results

Results for the selected impact categories are shown in Figure 3 for the cradle-to-grave life cycle of one hectare of high rainfall zone wheat production, with (the bar on the right of each set) and without use of nitrapyrin. For all the impact categories, the results have been normalized by the maximum value of the two, which is for the hectare treated with nitrapyrin for all impacts except GWP and marine eutrophication. The specific maximum values for each impact category are (per hectare):

- Climate change (GWP): 3,100 kg CO_2 eq
- Cumulative energy demand: 18,100 MJ
- Water withdrawals (ReCiPe): 17.8 m^3
- Terrestrial acidification (acidification potential, or AP): 9.53 kg SO_2 eq
- Freshwater eutrophication (eutrophication potential, or EP_f: 0.373 kg P eq
- Marine eutrophication (eutrophication potential, or EP_m): 55.2 kg N eq
- Photochemical oxidant formation (photo oxidant creation potential, or POCP): 5.39 kg NMVOC (non-methane volatile organic carbon)
- Ozone depletion (ozone depletion potential, ODP): 4.6E-05 kg CFC-11 eq

There are three primary observations from the figure:

- For most impact categories there is no significant difference between the with and without nitrapyrin cases
- Use of nitrapyrin clearly reduced GWP and marine (nitrogen-based) eutrophication
- Use of nitrapyrin may increase ozone depletion potential.

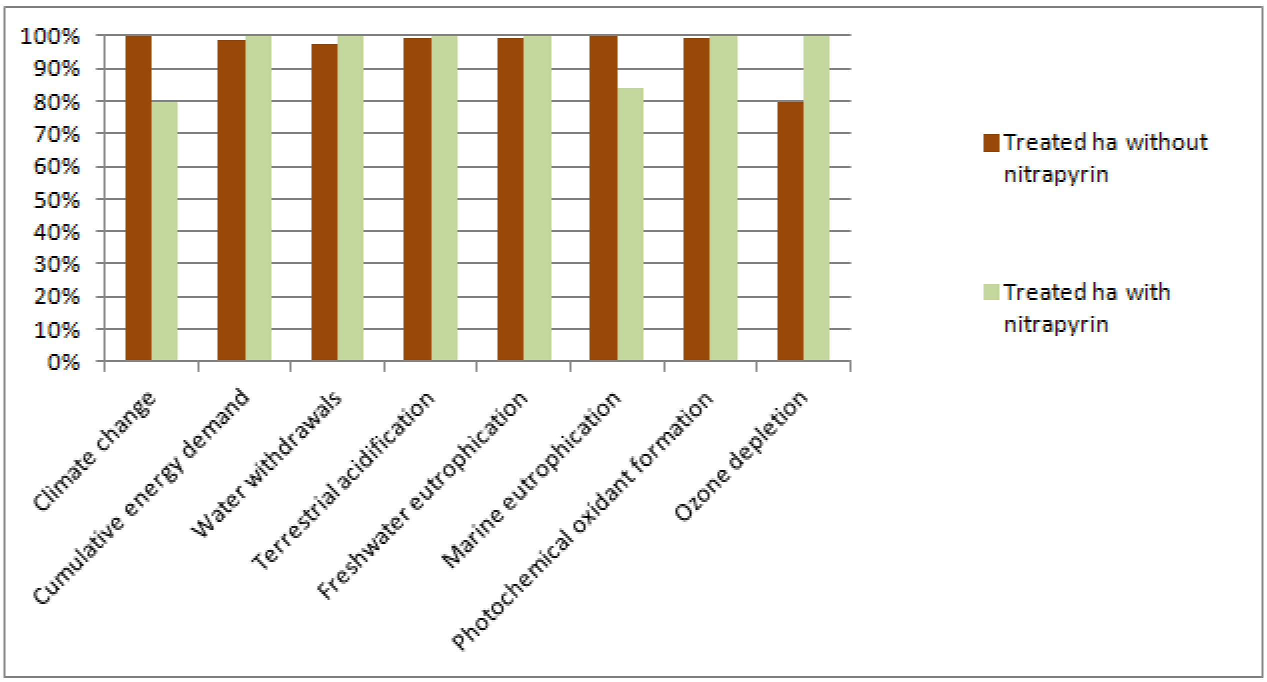

Figure 3. Comparison of potential impacts per hectare of HRZ wheat farming in Australia

The contributions of the different inputs to the full list of metrics considered are shown in Figure 4. For most metrics, the largest contributors are UAN production and single super phosphate production. Direct field emissions are significant for GWP and EP_m. Production of alachlor (a surrogate model for the use of clethodim on-farm) is the largest contributor to ODP. Production of nitrapyrin is a significant contributor only to ODP.

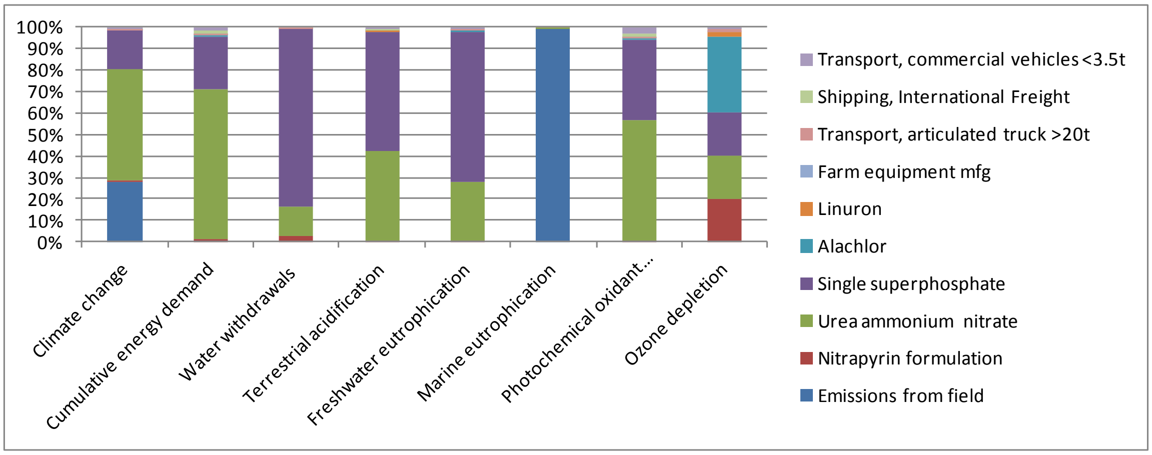

Figure 4. Contributions to potential impacts per treated HRZ hectare

4. Discussion

Use of nitrapyrin, which reduced modeled field emissions of N_2O by 65%, resulted in a GWP reduction of about 20%. The GWP of a treated hectare is 2470 kg CO_2eq/ha, which is 630 kg/ha less than the value for the hectare without nitrapyrin, 3100 kg CO_2eq/ha. About 22% of the GWP comes from field emissions of N_2O and only 0.5% comes from the nitrapyrin (11 kg CO_2eq/ha). The GWP benefit of nitrapyrin (648 kg CO_2eq/ha) is almost 60 times the GWP burden from manufacturing it. The largest contributor to GWP is the production of the UAN fertilizer, which contributes 52% of the GWP. The upstream burden of UAN production is more than twice as large as the impact from the N_2O emitted in the use of the product. There is also a significant contribution of 18% from phosphate fertilizer production and 6.5% from field emissions of CO_2 due to decomposition of the urea in the UAN. The production burden of UAN is still larger than the sum of the N_2O and CO_2 impacts from the use of the product in HRZ wheat farming. CO_2 emissions in UAN production are related to energy use, but N_2O is also a production reaction by-product, for which the emissions can be controlled by using different vent gas treatment technologies. The US EPA has compiled an excellent overview of the state of the art for N_2O abatement from nitric acid production (US EPA, 2010). Appendix C of the EPA report includes data on N_2O emissions (lb/ton) before and after implementation of technology changes at 45 plants in developing geographies. The average emissions of N_2O from the improved plants were 1.3 kg/MT, an 85% reduction from the emissions in the model for nitric acid production in the Australasian LCI dataset (which was unchanged from the European Ecoinvent value); the best technology could be 97% lower. We adapted the Australasian LCI model for nitric acid production by changing only N_2O emissions, neglecting any other inputs required to achieve this performance. Use of more current N_2O emissions technology in nitric acid production could lower the GWP of HRZ wheat farming by 16%, assuming that current nitric acid production in Australia is similar to that described in Ecoinvent. No other potential impacts were changed. This GWP impact is similar to that from using nitrapyrin, but is completely additive: using both more current nitric acid processes and nitrapyrin could combine to reduce GWP by 38%.

The decrease in marine eutrophication (N-load) by 16% with the use of nitrapyrin as shown in Figure 3 was directly due to the impact from the work by Wolt (2004), with the burdens created by supply of nitrapyrin being only 0.01% of the total. Although marine eutrophication is a significant issue for US agriculture, it may not be a significant issue for Australian agriculture (Geosciences Australia, 2013). The relevance of the reduction in marine eutrophication by use of nitrapyrin will depend on the region where it is used.

Figure 3 also showed increased ozone depletion potential with use of nitrapyrin in HRZ wheat farming. This is due estimated CCl_4 emissions from the chlorine production process at the plant of K2 PURE SOLUTIONS NOCAL LP in Pittsburg, California. This is a new facility (not owned or operated by Dow) and will need to publically report CCl_4 emissions, but data are not yet available for this facility (US EPA, 2012). When these data are available, it will be possible to determine if this potentially higher burden for nitrapyrin is real or not. The major contribution to ODP, production of alachlor, also has a high degree of uncertainty. ODP calculations are sensitive to small emissions of materials with high characterization factors. The use of alachlor as a surrogate for clethodim herbicide based on its structural similarity is reasonable for potential impacts such as GWP or cumulative energy demand, but is likely less so for ODP. The differences in ODP are likely not significant.

During the course of this project, there were also field studies in Managatang, Victoria, to measure the impact on yield of otherwise identically treated areas of wheat production, with and without use of nitrapyrin. A representative comparison using 2.5 L of eNtrench, 50 kg diammonium phosphate (DAP) and 40 L of urea ammonia nitrate increased the yield of wheat from 2.90 to 3.16 MT/ha, or about 9%. The quality of the wheat was similar for both treatments (11.7% protein without nitrapyrin; 11.8% protein with nitrapyrin). The yields, fertilizer inputs, and corresponding emissions were used to create models on a "per MT" basis. For most impact categories there was a small decrease between the cases with and without nitrapyrin, due to the 9% improvement in yield, although differences of this magnitude in LCA are often not considered significant. Use of nitrapyrin reduced GWP 29% and marine (nitrogen-based) eutrophication by 23%. Due to the lower input of fertilizer in the results per MT wheat in Managatang, many of the potential impacts were smaller in magnitude than those per typical treated hectare. This is demonstrated for GWP which was 3,100 kg CO_2eq/ha for the untreated typical hectare compared to 390 CO_2eq/ha for the reported field trial. Fertilizer production contributed 43% and field emissions contributed 44% of the GWP for the field trial without eNtrench; for the typical hectare these were 55% and 41%, respectively. Later field trials in Australia have shown that typically 5-7% increase in yield can be seen when nitrapyrin is used and all other factors are conducive to maximize the yield impact.

5. Conclusion

The potential benefits with respect to global warming potential (GWP) and marine eutrophication through the use of nitrapyrin in high rainfall zone (HRZ) wheat farming in Australia far exceed the burdens of these potential impacts created by its supply. Use of nitrapyrin could reduce global warming potential (GWP) by 20% and marine eutrophication by 16%, with burdens created by its supply of only 0.5% and 0.01%, respectively. There is no significant difference in the broad range of other potential impacts considered between a farmed hectare with and without use of nitrapyrin, with the possible exception of the ozone depletion potential. Production of nitric acid for urea ammonium nitrate fertilizer impacts GWP as much or more than the emissions from the field on its use. Reducing emissions from nitric acid production may provide a similar benefit in GWP as the use of nitrapyrin, and the potential benefits are independent and additive.

6. Acknowledgements

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Life cycle human exposure and risk assessment of pesticide application in Colombia: The example of potatoes

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ABSTRACT

Although the human health effects of pesticides have decreased, chronic health problems are still significant in many developing countries and emerging economies. In this project we examined the various exposure pathways of pesticide application over the whole life cycle of potatoes grown in Colombia. Exposure pathways included e.g. workers' exposure during pesticide preparation and application and consumer exposure by ingestion of the pesticide-treated crops. A dynamic model was developed for pesticide crop uptake and evaluated with measurements performed within a field trial in the region of Boyacá, Colombia. Pesticide concentrations were measured periodically in soil and potato samples from the beginning of tuber formation until harvest. The model was able to predict the magnitude and temporal profile of the experimentally derived pesticide concentrations well, with all measurements falling within the 90% confidence interval. Pesticides residues in potatoes were rather low and below health-based threshold values in the case investigated. However, the study was performed in an unusually dry year with smaller amounts of fungicides applied than in other years. Therefore, to study exposure and risk in the region under normal circumstances, an enquiry about pesticide use was conducted among 79 farmers of the region and the model applied to the application pattern of pesticides reported by the farmers. Results show that substitution of a few active ingredients could lower consumer exposure significantly. With regard to workers' health, dermal exposure was found to be enhanced in only several cases. Several measures are suggested to lower workers' exposure. Finally, human intake of pesticides was quantified and compared throughout the life cycle of potatoes. Cumulated intake fractions of consumers' ingestion and farmers' dermal exposure were comparable, but individual doses for farmers were much higher and above no-effect-levels. This highlights that individual risk assessment studies are needed in addition to LCA, which typically cumulates intake.

Keywords: pesticides, toxicity, plant-uptake model, farmers' exposure

1. Introduction

Pesticides can help to avoid harvest losses and increase yields, but may come at the expense of health effects if not correctly applied. Although the human health effects of pesticides have decreased significantly in industrialized countries, misuse of pesticides in developing countries is still problematic. Possible factors contributing to these effects include the application of old products with high persistence and toxicity and missing or insufficient protection of workers during pesticide application and use. Consumers, farmers and authorities are interested in understanding and ultimately mitigating the life-cycle environmental and health impacts related to the use of pesticides.

The goal of this project was to examine the various exposure pathways of pesticide application over the whole life cycle of potatoes grown in Colombia. Potatoes were chosen for this study as it is the vegetable that is consumed most (e.g. per capita consumption in Columbia of 42 kg/capita/y), involves small-scale farming and is the crop with high pesticide use in Colombia.

2. Methods

We collected data, performed experimental studies and developed models to quantify the magnitude of exposure to pesticides from the preparation of pesticide solution, the application on the field, as well as post-harvest consumer exposure due to ingestion of potatoes.

2.1. Crop uptake model and field measurements

A dynamic model for uptake of pesticides in potatoes was developed and evaluated with measurements. The model takes into account the time between pesticide application and harvest, the time between harvest and consumption, the amount of spray deposition on soil surface, mobility and degradation of pesticide in soil, diffusive uptake and persistence due to crop growth and metabolism in plant material, and loss due to food processing like cleaning, washing, storing, and cooking. Intake fractions were calculated according to Equation 1.

$$iF_{consumers} = PF \cdot C_{potato} \cdot Y \cdot n / M \qquad \text{Eq. 1}$$

Where iF is the intake fraction, PF the processing factor (loss of pesticide by processing, e.g. washing or cooking), C_{potato} the concentration in the potatoes, Y the yield, n the number of people exposed, and M the mass of pesticide applied.

A field trial was performed on a farm in the region of Boyaca. This trial included periodical measurements of pesticide concentrations in soil and in the potato samples from the beginning of tuber formation until harvest. Application patterns (times, amounts and types of pesticides applied) were recorded.

For a complete documentation of the model and the results of the field experiments see Juraske et al. (2011).

2.2. Farmers' exposure

Dermal and inhalation exposure experiments of pesticides in potato cultivation systems in Colombia were carried out. Farmers' exposure during preparation and knapsack-sprayer application of pesticide was quantified with tracer experiments and the whole-body dosimetry methodology (Lesmes-Fabian et al. 2012). Intake fractions of the farmer were calculated according to Equation 2.

$$iF_{farmer} = M_{dermal} / M \qquad \text{Eq. 2}$$

Where iF is the intake fraction, M_{dermal} the mass of pesticide deposited on the skin of the farmer applying the pesticides, and M the total mass of pesticide applied.

2.3. Survey

To understand and calculate exposure in the region, a survey was conducted. Pesticide application data was collected through interviews with 79 farmers in the region of Boyacá, covering a total cultivation area of 82 ha. This data was then combined with the models developed to estimate exposure.

3. Results

The survey showed that 22 different active ingredients were used with an average of 9.8 kg/ha. 90% of total mass applied due to only 4 pesticides (carbofuran, carbosulfan, mancozeb, and methamidophos). Figure 1 shows the distribution by mass of these pesticides.

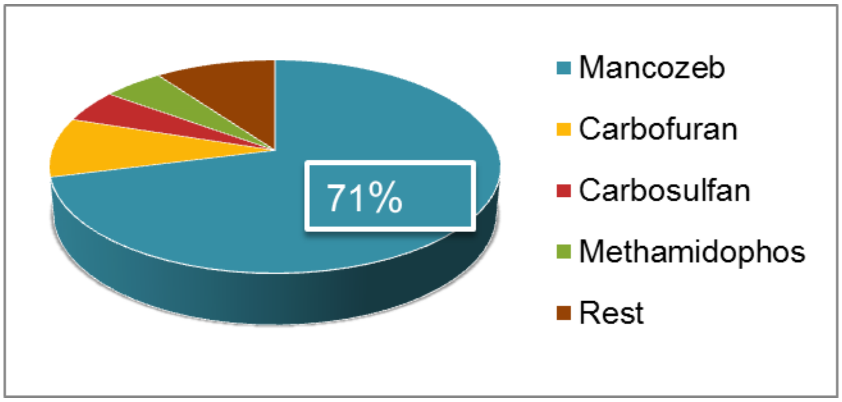

Figure 1: Share of pesticide masses applied in the study area

Modeled data for crop uptake was in good accordance with experimental data (Juraske et al. 2011). Cumulative intake fractions of consumers through ingestion of potatoes were $iF_{consumers} = 10^{-4}$ for cooked potatoes consumed directly after harvest (no cleaning/washing), $iF_{consumers} = 10^{-6}$ for cooked potatoes consumed 3 months after harvest (no cleaning/washing) and $iF_{consumers} = 10^{-5} - 10^{-7}$ for cooked potatoes that were previously cleaned and washed. Pesticides residues in potatoes were rather low and below health-based threshold values in the case investigated. However, the study was performed in an unusually dry year with smaller amounts of fungicides applied than in other years.

Table 1: Exemplary results of applied doses mean concentrations (n=3) and coefficient of variation of pesticides detected in soil (A) and pesticides applied but not detected in soil (B) in one field trial (Juraske et al. 2011). Samples taken at average tuber depth (12 cm).

(A)

	DDT	DDD	DDE	Carbofuran	Chlorpyrifos	m-parathion
Dose (kg ha^{-1})	-	-	-	-	0.435	-
Conc. (mg kg^{-1})	0.29	0.13	0.25	0.5	2.11	0.18
CV (%)	19	26	27	25	32	76

(B)

	chlorothalonil	cymoxanil	glyphosate	mancozeb	metamidophos	paraquat
Dose (kg ha^{-1})	0.54	0.08	0.14	0.66	0.55	0.08
Conc. (mg kg^{-1})	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.

To study exposure and risk in the region under normal circumstances, an enquiry about pesticide use was conducted among 79 farmers of the region and the model applied to the application pattern of pesticides reported by the farmers. Questions were related to:

- Household description
- Potato field description
- Pesticide and fertilization management
- Occupational hygiene (use of personal protective equipment)
- Health problems related to the use of pesticides

Results show that substitution of a few active ingredients could lower consumer exposure significantly. However, in spite of some cases in which too much pesticide was applied too late before harvest, concentrations in potatoes were mostly low at harvest (below maximum residue levels MRLs). By contrast, farmers' exposure during application was significant, particularly dermal exposure. Dermal exposure depended on type of work clothing, cleaning of the application equipment, and application frequency. Although cumulated intake fractions of consumers' ingestion and farmers' dermal exposure were comparable, individual doses for farmers were much higher and above no-effect-levels. This highlights that individual risk assessment studies are needed in addition to LCA, which typically cumulates intake across all people exposed.

4. Conclusion

This study investigates various pathways for human health effects from pesticide exposure and, hence, focuses only on one impact category. While this was fine for the scope of the analysis in this case, also other impact categories than human toxicity need to be addressed when performing a complete LCA study.

The system showed to be rather resilient in terms of uptake of pesticide in the potatoes. Only in very few cases elevated residues were found, while for the majority of applications no health impacts from potato consumption are to be expected. By contrast, pesticide exposure of farmers from the application of pesticides was elevated and health impacts have been reported. The level of human health risk was especially for the pesticides like metamidophos. Exposure and health effects could be lowered by substituting the most toxic active ingredients, avoiding unnecessary application of pesticides, wearing appropriate protection clothing made of thick fabric and covering the whole body and cleaning all spill residues on the sprayer tank before starting the application.

The results of this study illustrate the importance of considering farmers/workers' exposure, which is often neglected in common LCA studies. However, it also shows the shortcomings of the LCA approach. In spite of the differences in individual exposure, LCA would have come to the conclusion that exposure of consumers and the farmers are comparable, as typically LCA does not consider individual but cumulative exposure. One farmer takes in the same amount of pesticide as a multitude of consumers together via potato consumption, resulting in similar intake fractions according to equations 1 and 2. However, the farmer is much more likely to exhibit health effects, as his individual dose of pesticide is much higher. This shows that risk assessment studies are always needed in addition to LCA, in order not to miss important effects.

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Model for uptake of pesticides in potatoes (and other crops): www.dynamicrop.org

LCA is only relative – Experiences from the quantification of overall dispersions around aquaculture LCI results

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ABSTRACT

As part of an evaluation of European seafood imports from Asia, overall dispersions were determined for a wide range of processes. Once propagated into LCI results using Monte Carlo (MC) simulations, large absolute uncertainties around results made conclusions difficult to reach. In response, the results were reevaluated, only considering relative uncertainties related to different hypotheses. This approach allowed for dispersions related to more than one of the products under study to be assumed the same, producing correlated samples. These results could, in turn, be tested using paired T-tests which allow for greater statistical power. Greater statistical power means that the resolution of comparisons is improved and the risk of reaching flawed conclusions reduced. Among other bias related to absolute results, the present research presents many contending arguments for only comparing LCI and LCA results in relative terms.

Keywords: LCI, uncertainty, dispersion, Monte Carlo, significance, statistical test

1. Introduction

Quantifying uncertainties related to life cycle inventory (LCI) results has long been desired in the field of life cycle assessment (LCA). To date, however, most estimates of uncertainties have been derived using a pedigree approach (Weidema and Wesnaes 1996), based upon expert judgment (Röös et al. 2010), or focused on a few key parameters (Middelaar et al. 2013), resulting in manageable ranges. In the few studies where parameters have been more extensively defined (e.g. Gregory et al. 2013), comparisons of products or services are often inconclusive. This is due to the large dispersions related to LCI results. Meanwhile, results are commonly communicated as point values, with little to no indication of the confidence behind any conclusive outcomes. Supporting LCI outcomes with information on its uncertainty and statistical testing is therefore needed.

The first step towards implementing statistical tests to LCI results is to identify all relevant sources of dispersion. In the present research, the *overall dispersions* defined in Henriksson et al. (2013) were adopted. Overall dispersions are the sum of inherent uncertainties (inaccuracies in measurements which may be reduced by additional research), spread (variability resulting from averaging) and unrepresentativeness (mismatch between the representativeness and application of data) (Henriksson et al. 2013). Secondly, data need to be collected and assembled following a representative sampling technique. However, as data collection commonly is the most resource demanding stage of any LCA, and LCA is data intensive, most studies source their data opportunistically. This results in data often representing very small sample sizes (n), surveys carried out using snowball sampling, geographically limited areas, temporally limited timespans or otherwise biased samples. Thirdly, a propagation method is needed to aggregate the unit process dataset into LCI results. In the field of LCA, Monte Carlo (MC) simulations with random virtual sampling is the most common propagation method (Lloyd and Ries 2007). While MC simulations of large datasets require relatively extensive computing, advances in hardware and software today allow for a great number of iterations to be produced from extensive LCI databases within a reasonable time using a standard personal computer.

When all sources of dispersion are taken into account, the resulting *absolute uncertainties* often become very large, making it difficult to draw conclusions in comparative studies (Henriksson et al. 2014). Common when comparing LCI results, however, is that many of the sources of dispersion are positively correlated with each other (Hong et al. 2010). In LCI studies this may refer to dispersions related to emission models (e.g. CH_4 estimates from diesel combustion), emissions from land-use change, or other processes which are shared amongst the production systems compared. Therefore, in comparative studies, only *relative uncertainties* are of importance and all shared dispersions can be assigned the same random samples. E.g. if an LCA evaluates two toasters, the uncertainties related to electricity generation can be assumed identical and only the uncertainty related to the amount of electricity used and the production of the toasters should be considered. Thus, only dispersions unique to either of the products under study, the relative uncertainties, are considered.

The types of statistical tests that can be implemented to test LCI results depend upon the characteristics of the data. Parametric statistics, for example, allow for more statistical power but assume equal variances and a type of probability distribution (Table 1). The probability distribution assumed by the most commonly used statistical tests (e.g. the T-test) is the normal distribution, but other test exist and distributions can also be transformed (e.g. from a lognormal to a normal distribution). In the meantime, non-parametric tests which do not require data to conform to a specific probability distribution offer more robustness, but an increased risk of committing a type II error (see Table 1). Implementing the wrong type of test can consequently result in flawed conclusions. For example, a novel biofuel might be deemed as equally polluting as diesel, while it in fact has significantly lower emissions. If only relative uncertainties are considered, the outcomes of each individual MC run for the product systems compared are correlated with each other (Wilcoxon 1945; Zimmerman and Zumbo 1993). This is referred to as correlated samples and implies that a paired statistical test should be implemented, where only the difference between product A and product B (A-B) is considered for each MC run. Paired statistical tests offers increased statistical power and are often used for e.g. testing groups of patients before and after a drug is administered (Zimmerman and Zumbo 1993).

Table 1: Common statistical terms used throughout this manuscript

Parametric statistics	Statistics of data of an assumed type of probability distribution; offers more statistical power and less chance of type II errors
Type II error	Failure to reject a false null-hypothesis
Statistical power	The probability of rejecting the null-hypothesis when the alternative hypothesis is true
Non-parametric statistics	Statistics of data with no characteristic structure; offers more robustness and less chance of type I errors
Type I errors	An incorrect rejection of the null-hypothesis
Statistical robustness	The ability to evaluate data of no characteristic structure, data affected by outliers, or data where variances are not equal

In the present research we evaluated the advantages of only considering relative uncertainties when comparing LCIs and challenged the use of absolute values. In order to do so, we worked on an inventory dataset describing Asian aquaculture. The data were collected as part of the SEAT project (www.seatglobal.eu) for several important aquaculture systems in Asia. In the present study, however, we only used the example of carbon dioxide emissions from the production of one tonne of tilapia fillets from two different Chinese farming systems, conventional and integrated with pigs, as an example. Our hypothesis was that "*fish integrated with pigs would have lower emissions than conventional systems as part of their feed is provided by the pig manure*".

2. Methods

Primary data collection started in 2010-2011, building upon a random sample of 200 fish farms (Murray et al. 2013). Based upon the outcomes of this sample, additional primary data were collected in 2012-2013 on supporting processes, including feed mills, hatcheries, processing plants, etc. The primary dataset was also supported by an extensive review of secondary data, detailing many of the supporting services (e.g. electricity production, transportation, etc.). For these data, aggregation of dispersion parameters was conducted using the spreadsheet supplied as Online resource to Henriksson et al. (2013) (also available at cml.leiden.edu/software/software-quanlci.html). For all other supporting processes the ecoinvent v2.2 database was consulted, relying upon default inherent uncertainties and the pedigree approach presented by Frischknecht et al. (2007). A full review of the data used and modelling choices adopted is available in Henriksson et al. (2014). Modeling and propagation was conducted in the CMLCA software v5.2 (available at cmlca.eu). For each farming system alternative 1 000 iterations were conducted. Probability density functions were evaluated in EasyFit v5.5 (mathwave.com), using the Anderson-Darling test to determine the distribution of data. Statistical tests were conducted using Wilcoxon matched-pair signed-rank test in SPSS v.21.

3. Results

The outcomes from the correlated and the non-correlated sampling were largely identical, apart from some random sampling differences. Between the two, integrated farms performed slightly better than the conventional farms (Table 1). The relative differences between the two data ranges (conventional-integrated) were not normally distributed.

Table 2: LCI results for carbon dioxide emissions from the production of one tonne of tilapia in either of two different Chinese tilapia farming systems.

	Conventional	Integrated
Average	3027	2684
Median	2805	2530
Standard deviation	1226	953

When compared on a MC run basis, two products with equal emissions would each be expected to yield the higher emissions 50% of the time. Meanwhile, the integrated farms came out as having higher emissions than conventional farms only 43% of the runs when absolute results were regarded, and 39% when relative results were regarded. The difference in the dependently sampled results could also be determined as highly significant ($p<0.001$) using the non-parametric Wilcoxon matched-pair signed-rank test. When visualized for a small selection of MC runs (Figure 1), the correlation between the samples in the dependent sampling becomes evident. Where only relative uncertainties are considered (dependent sampling) the samples generally come out closer to each other than when absolute uncertainties are considered (independent sampling). This as the results rely upon the same LCI matrix, where e.g. the assumed emissions from electricity generation and transportation in China remain the same for both production chains in each MC run.

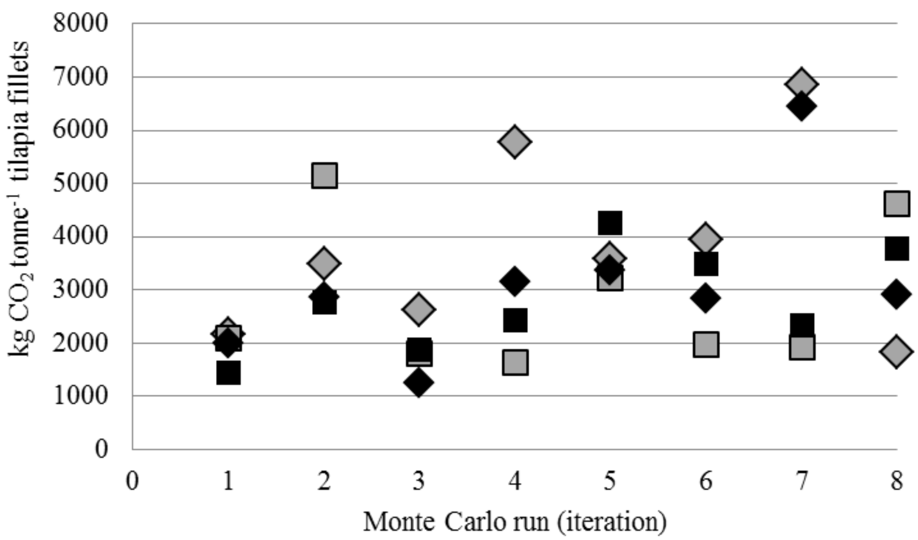

Figure 1: Illustrative example of a selection of outcomes using correlated (black) and non-correlated (gray) sampling for carbon dioxide emissions from conventional (diamonds) and integrated (squares) tilapia farming in China.

4. Discussion

Given the definition of the overall dispersions evaluated here, the ranges of results give an indication of the *absolute uncertainties* underlying results. Meanwhile, if two production chains which experience many shared sources of dispersions are to be compared, only relative uncertainties need to be considered. Only considering relative uncertainties improve the resolution of comparisons and also allow for the implementation of more powerful paired statistical tests. The field of LCA should therefore amend the practices of most other sciences, where results are only compared to an alternative or control (benchmark), working towards a predefined hypothesis.

Although often used in LCA studies instead of great or large, the word significant in the scientific realm implies that a statistical test has been performed. Meanwhile, only a handful of LCA studies have so far implemented statistical testing to support their conclusions (Canter et al. 2002) and the role of statistical tests in the field of LCA is still vaguely defined. Embracing statistical concepts would, however, contribute towards more solid LCA practices. Essential for this will be the understanding that statistical tests are only ever as strong as the sampling framework they build upon. Sampling frameworks therefore need to promote random unbiased samples, where any remaining bias is communicated so that it can be corrected for. Moreover, the probability density function of LCI outcomes should be characterized and transformed when needed, in order to allow for parametric tests. Otherwise there is a risk that type II errors are committed.

If the approach proposed here is adopted, modeling options may also need to be considered. For example, if a generic process for “Combustion of diesel in generator” is used, rather than defining its emissions as part of each process using a generator, dependent sampling becomes possible. Similarly for waste flows, where e.g. a waste processes may be created for each kg of nitrogen applied to agricultural fields. While the uncertainties for these more generic processes might be larger, they are less relevant from a relative point-of-view. Dependent sampling is also beneficial when uncertainties around characterization factors are considered. In some cases, such as freshwater ecotoxicity, uncertainties may range an order of magnitude (Henderson et al. 2011). In such cases, adopting dependent sampling could greatly improve the resolution of comparative studies.

Further efforts are needed to develop the ideas presented here. For example, the strong influence of sample size in most statistical tests. When results are resampled, as in MC simulations, the sample size can be changed at the click of a button. This means that a null-hypothesis might not be rejected (no significant difference) at 100 iterations, but at 1 000 iterations. This rather arbitrary feature has even been addressed by implementing sequential stopping boundaries, where MC tests will stop once a sufficient number of runs have been produced to achieve a certain significance level (Fay et al. 2007). Future research should therefore explore how to deal with such resampling risks in the field of LCA (Gandy 2009) and evaluate statistical tests building upon the cumulative distribution function, such as the Kolmogorov–Smirnov test.

5. Conclusion

LCAs presenting results as point values may be flawed when concluding that one product is better than another while no significant difference exists (type I error), or to conclude that two results are indifferent from each other while they actually significantly differ (type II error) when dispersions are quantified as yet. Either of these false outcomes may result in flawed decisions, and only by quantifying the dispersions around LCI results can statistical tests be implemented and the confidence behind conclusions communicated. In doing so, dependent sampling only regarding relative uncertainties offers many advantages. This, however, requires studies to work towards a hypothesis, in order to avoid multiple comparison problems. Multiple comparison problems refer to the elevated chance of finding significant trends by chance if a large number of comparisons are made. Amending these recommendations will allow LCA to become a much more scientifically robust tool, rather than an exploratory framework. Conclusively, we argue that LCI and LCA results should only ever be seen as relative, and discourage comparisons of absolute results across studies.

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Water-use and impact-weighted water footprints – methodological approach and case study for two Austrian milk production systems

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ABSTRACT

Freshwater is a scarce resource in many parts of the world. Besides the quantitative restriction, a potential contamination of water resources ranks high on the agenda of environmental concerns in many regions. The objective of this paper is to describe methods for impact-weighted grey water footprints (WFs), derived from grey water-use (WU) estimates. Furthermore, total WU (blue surface- and groundwater, green evapotranspiration water from precipitation and virtual grey dilution water) and WFs (impact-weighted blue and grey water footprints) are calculated based on life cycle assessment (LCA) principles for two Austrian milk production systems (PS). We identified two approaches for impact-weighted grey water results: (1) a regional catchment area-based approach and (2) a local approach. It is shown for the milk PS that differences in management result in different virtual grey WU with high importance for the overall water demand. Analogously management- and precipitation-related impact-weighted grey WFs completely dominate the overall WF results.

Keywords: water-use, water footprint, nitrate emissions, grey water, impact-weighting

1. Introduction

Freshwater is a scarce resource in many parts of the world. Besides the quantitative restriction, a potential contamination of water resources ranks high on the agenda of environmental concerns in many regions. In recent years, scientific studies have started to analyze factors influencing water-use (WU), the resulting impact on water-resources and water quality induced by diverse production systems and management strategies for comparable food products. Earlier studies on amounts of "virtual water" (see e.g. Allan et al. 2003) or on different types of so-called "water footprints" (WFs; see e.g. Hoekstra et al. 2011) implemented methodologies with lower resolution to analyze and illustrate hydrological aspects in a broader context: the concepts were primarily developed to show effects of WU in all stages of supply chains for nations, companies or products and to illustrate virtual global water flow.

Previous studies focused on food production's water consumption only, but did not or not fully account for the effects of production on water quality. From these it was often concluded that high-output production results in a lower water demand per unit of product output. Contrarily, a few recent studies using refined methodology demonstrate effects of different production systems on water aspects with a focus on water quality and grey water (i.e. virtual water demand to dilute the main emitted pollutant in a specific water body) on a product level: For example, Ercin et al. (2012) compared soy production from different countries and diverse production methods and found that shifting from non-organic to organic farming may reduce the grey water related to soybean cultivation by 98%. Franke and Mathews (2013) found that organic farming practices for Indian cotton showed a five times smaller grey water demand than for conventional production, while having comparable land productivities as in conventional farming. However, these concepts are based on WU at an inventory level, but the water demand is not weighted by its impact on a water system which could influence a regional or local WF result. In contrast to a global warming potential (carbon footprint), which provides a measure for an impact on global warming with globally equal impact factors for the different greenhouse gases, impacts of WU vary spatially and temporally and because of the related methodological difficulties, they are hardly incorporated in WF studies. Opposed to the unweighted WU, a few impact assessment methods were created in recent years, e.g. the water stress index (WSI; Pfister et al. 2009). The latter was developed for blue water (ground- and surface-water used) only and characterizes the impact relevance of regional water consumption (for midpoint modelling). Regionally varying stress indicators are used, ranging from 0 (no water stress) to 1 (extreme water stress) for more than 11'000 globally differentiated watersheds in order to calculate product water footprints.

The objectives of this paper are: (1) to propose a method that is based on life cycle assessment (LCA) principles and LCA data and fully accounts for all kinds of WU (blue, green and grey water) and also for impact-weighted WFs (red and impact-weighted grey water); (2) to describe characterization methods which address impact-weighted grey WFs. (3) Furthermore, to demonstrate the implementation of this method, WU results (for blue, green and grey water) and WFs (red and impact-weighted grey water) are calculated per kg milk for two typical different Austrian milk PS.

2. Methods

2.1. Proposed method, scope and system boundaries

This contribution presents a method for water-use (WU) and water footprint (WF) estimates for agricultural products, which is based on LCA principles and accounts for all relevant life cycle-related inputs for the inventory level. This includes: (1) blue water, i.e. surface or groundwater used during production processes (e.g. in housing or for irrigation); (2) green precipitation water which is evapotranspirated by plants and soil; (3) grey water, i.e. the amount of water needed to dilute contaminants in affected freshwater. For the impact-weighted water footprints, the proposed method covers the so-called red WF, i.e. an impact-weighted blue WF, (see e.g. Pfister et al. 2009) and introduces an impact-weighting for grey water, which has not been covered in previous contributions (see e.g. Launiainen et al. 2014, Chenoweth et al. 2013). Deriving red water through a weighting step for blue water is meant to taking into account the regional stress on freshwater resources that is related to a WU. In analogy, grey WU results are impact-weighted in order to represent the water quality stress: any contamination of water not only leads to an increase in the grey WU of a product, but exerts a particular impact if an affected water body's quality is already impaired, increasing the amount of water needed to dilute the contaminant to acceptable contents. The NO_3-contamination of groundwater related to agricultural production may serve as an example for this: The NO_3-content in water bodies is mainly affected by the amount of precipitation water and flowing water for groundwater and surface water bodies, respectively. Additionally, it is influenced by the amount of NO_3-N emitted into the water systems, depending on the cultivation management and on further NO_3-loads e.g. from deposition. The proposed impact-weighting does not only account for the specific grey water demand of an assessed product, but also includes the effect of regional NO_3-loads on groundwater. The impact-weighting factor depends on the groundwater NO_3-content, which is assumed to be influenced by long-term cultivation management and related NO_3-loads. For the local approach, constant long-term conditions are assumed, which are reflected in the current state of the respective water bodies.

Green water, which is included at the inventory level (WU method), is not considered for the impact-weighted level (WF method) any more. This green water is not comparable to blue water, as it is assumed not to contribute to water scarcity from a water management perspective, and its impact is related to the indicator land use (Ridoutt and Pfister 2010). Contrarily, other authors argue that green water is also limited, can be substituted by blue water and may affect blue water availability (Jefferies et al. 2012, Berger and Finkbeiner 2012, in: Chenoweth et al. 2013); consequently, we include green water on the inventory level.

Figure 1 illustrates water types according to different methods and results included in this paper for WU (inventory level) and for WFs (impact level), i.e. red water according to Pfister et al. (2009) and an impact-weighted grey water footprint affected by the background concentration of the substance to be considered (here: nitrate).

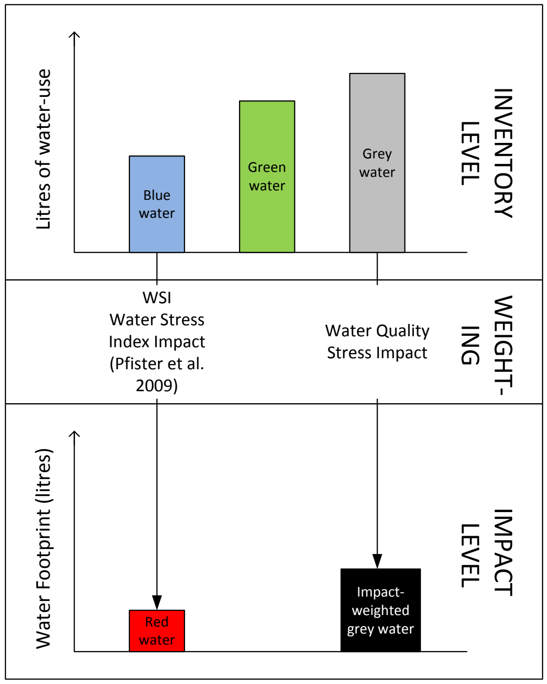

Figure 1: Water types covered in our method for water-use estimates and a product water footprint.

The concept of WU is based on Hoekstra et al. (2011) concerning blue, green and grey water but uses other data sources and calculation procedures for quantification. Table 1 gives an overview of the calculation procedures and of some of the algorithms provided in this paper for WU and WF calculations at different stages in the life cycle for the case of milk production. System boundaries were defined to include the most important processes requiring water, such as the supply of input factors relevant for the production of feedstuffs, evapotranspiration of plant feedstuffs, water directly used for livestock husbandry and for milk processing in the dairy (Table 1). Generally, WU (water demand) is calculated as the product of activity data (e.g. amounts of mineral nitrogen fertilizers to produce one unit of feedstuff) and water demand factors per unit (e.g. liters (L) of blue water needed per kg nitrogen fertilizer produced). While some WU estimates such as for blue drinking and cleaning water used in housing represent gross water demands, the red WF results describe further weighted net losses of water from a local system (e.g. irrigation water lost by evapotranspiration or water contained in products leaving the local system). Grey water is generally not lost from a local system, but is a virtual water demand which is accounted for in the water balance to include effects of water pollution on water demands.

Table 1: Different water types and algorithms provided for water-use and water footprints for different elements of a milk production system.

	Water types covered at inventory level	Equa-tion	Water types covered at impact level	Equa-tion
Cultivation of feedstuffs	$\text{Blue}_{\text{irrigation}}$	1	$\text{Red}_{\text{irrigation}}$	2
	$\text{Green}_{\text{evapotranspiration}}$	3		
	$\text{Grey}_{\text{leaching}}$	4	Impact-weighted $\text{grey}_{\text{leaching}}$	5 / 6
	$\text{Grey}_{\text{energy-use}}$		Impact-weighted $\text{grey}_{\text{energy-use}}$	
Production of mineral fertilizers/pesticides	$\text{Blue}_{\text{fertiliser-processing}}$		$\text{Red}_{\text{fertiliser-processing}}$	
	$\text{Grey}_{\text{energy-use}}$		Impact-weighted $\text{grey}_{\text{energy-use}}$	
Industrial processes for feed production and in the dairy	$\text{Blue}_{\text{feed-processing}}$		$\text{Red}_{\text{feed-processing}}$	
	$\text{Grey}_{\text{energy-use}}$		Impact-weighted $\text{grey}_{\text{energy-use}}$	
Livestock husbandry	$\text{Blue}_{\text{housing}}$		$\text{Red}_{\text{housing}}$	
	$\text{Grey}_{\text{energy-use}}$		Impact-weighted $\text{grey}_{\text{energy-use}}$	
Transports (between & within all elements) and trade	$\text{Blue}_{\text{cleaning}}$		$\text{Red}_{\text{cleaning}}$	
	$\text{Grey}_{\text{energy-use}}$		Impact-weighted $\text{grey}_{\text{energy-use}}$	
Dairy processing	$\text{Blue}_{\text{processing}}$		$\text{Red}_{\text{processing}}$	
	$\text{Blue}_{\text{cleaning}}$		$\text{Red}_{\text{cleaning}}$	
	$\text{Grey}_{\text{energy-use}}$		Impact-weighted $\text{grey}_{\text{energy-use}}$	
	($\text{Grey}_{\text{wastewater}}$)		(Impact-weighted $\text{grey}_{\text{wastewater}}$)	

2.2. Blue water use for irrigation – methodological aspects and input data

For cultivation of feedstuffs, regionally varying amounts of blue water have to be used for irrigation ($Blue_{irrigation}$) to compensate for a lack of precipitation. Equation 1 [Eq.1] quantitatively describes this WU (liters of blue water for irrigation per kg milk).

$$Blue_{irrigation} = \sum_{i=1}^{n} Feed_{i,r,m} * I_{i,r,m} = \sum_{i}^{n} Feed_{i,r,m} * irr_{i,r,m} * ET_{i,r,m} \qquad \text{Eq.1}$$

where $Feed_{i,r,m}$ is the amount of different feedstuffs i from specific regions r, produced with management practices m to yield one unit (kg) of milk, multiplied with the feedstuffs' specific irrigation water demands $I_{i,r,m}$ (specific for feedstuffs/crops i, for regions r and for management practices m; in L kg^{-1} DM). $ET_{i,r,m}$ describes the evapotranspiration water demand (in L) for selected Austrian feedstuffs i as derived from the EPIC model (Schmid 2011, Asamer et al. 2011; see Table 2). The term $\%irr_{i,r,m}$ represents the proportion of irrigation water from overall water for ET plus recharge water amounts for crop i in region r for management practices m. Due to mostly sufficient precipitation for evapotranspiration, blue water is commonly not used for irrigation feedstuff production in Austria, where only a small proportion of cropland is irrigated. Most feed producers do not have the infrastructure for irrigation. Consequently, a proportion ($\%irr_{i,r,m}$) of only 1 % and 2 % of overall evapotranspiration water $ET_{i,r,m}$ was estimated for Austrian grain and corn production, respectively. Siebert and Döll (2010) show a comparable percentage of blue water from overall virtual water for crops from other regions which mainly export crops to Austria. Hence, we assumed the same values for $\%irr_{i,r,m}$ for imported cereal grains, maize and soybeans.

We used mass allocation for co-products, e.g. 1 kg of soybean cake shows an equal evapotranspiration water demand as toasted soybeans or extracted soybean meal (see Table 2).

Table 2: Water required for evapotranspiration for selected concentrate feedstuffs (liters per kg DM).

Feedstuff	$ET_{i,r,m=Austria}$ m = Organic / Conventional	Feedstuff	$ET_{i,r,m=Austria}$ m = Organic / Conventional
Triticale & wheat	471.5 / 480.8	Lucerne meal	502.0 / 528.7
Barley	467.4 / 490.9	Wheat bran	471.5 / 480.8
Maize (corn)	505.7 / 544.9	Soybean cake	477.5 / 495.1
Peas	455.2 / 448.6	Extracted soybean meal	477.5 / 495.1

2.3. Irrigation-related red water footprint

For characterizing the irrigation-related water footprint, we applied the Water Stress Index WSI (Pfister et al. 2009) to our inventory results $Blue_{irrigation}$ for the feedstuffs $Feed_{i,r,m}$. We assumed that all feedstuffs for the milk production were cultivated in Austria or in the neighboring countries (Hungary, Slovakia, Slovenia) and Croatia or Romania, which all belong to the water catchment basin of the Danube. Hence we multiplied all blue water results from the inventory level with a WSI-factor (WSI_r) of 0.0689 for this water catchment basin according to Pfister et al. (2009). As an exemption, imported soybeans for extracted soybean meal are assumed to be imported mainly from Brazil, Argentina and USA; on average, we assumed a WSI-factor of 0.012 for imported soybeans (Pfister et al. 2009). See equation 2 to for the detailed calculation:

$$Red_{irrigation} = Blue_{irrigation} * WSI_r = \sum_{i=1}^{n} Feed_{i,r,m} * I_{i,r,m} * WSI_r \qquad \text{Eq.2}$$

2.4. Green water (precipitation) use – methodological aspects and input data

Green water was estimated at the inventory level as the difference between the overall water demand for evapotranspiration as derived from the EPIC model and the amount of irrigation water. Equation 3 describes this green water demand from precipitation which results from evapotranspiration and is usually removed from local water cycles. With the proposed method green water is accounted for at the inventory level; it is also limited and can be substituted by blue water. Additionally, we estimated a yield-related loss of green water from reduced precipitation in the case of preceding clearing of tropical forests (based on data from Avissar and Werth 2005). This loss of green water from a regional water cycle is relevant for a part of soybeans from South America.

$$Green_{evapotranspiration} = \sum_{i=1}^{n} Feed_{i,r,m} * \left(ET_{i,r,m} - I_{i,r,m}\right) = \sum_{i=1}^{n} Feed_{i,r,m} * \left(ET_{i,r,m} * (1 - \%irr_{i,r,m})\right) \quad \text{Eq.3}$$

2.5. From grey water-use to the impact-weighted grey water footprint

The concept of grey water related to crop cultivation is based on Hoekstra et al. (2011) but was refined: First we confirmed nitrate to be the most relevant substance in terms of limiting ground water quality for most of Austria (Austrian Environment Agency 2007). Furthermore, we changed the constant 10 % nitrogen (N) assumed by Hoekstra et al. (2011) to be leached as nitrate to proportions for average conventional production, which were found in a meta-analysis covering 127 studies (Kolbe 2002). As a result of the meta-analysis, 26.6 % and 11.0 % of applied N for conventional management were calculated to be lost as nitrate from arable land and permanent grassland, respectively. The parameter $\%leaching_{i,r,m}$ in equation 4 represents the proportion of lost NO_3-N in connection to $N_{applied\ i,r,m}$, i.e. the amount of N applied per kg of feed from fertilizers including available N from biological N-fixation and deposition of gaseous N-losses from livestock husbandry (see Hörtenhuber et al. 2010 and Hörtenhuber et al. 2011 for activity data on N loads and N applied as well as N-emissions for selected feedstuffs). As the term ($N_{applied\ i,r,m}$ * $\%leaching_{i,r,m}$) is given in kg NO_3-N, it has to be converted into NO_3 (nitrate) by division (0.2259).

In contrast to previous literature sources, this contribution does not include a "regional background concentration" of nitrate in affected water bodies for calculation of grey water. Hence, for "$Grey_{leaching}$" per kg milk, i.e. the amount of virtual grey water which is needed to dilute pollutants in fresh water, we used a maximum tolerable limit for drinking water quality ($C_{max\ NO3}$) of 45 mg (0.000045 kg) NO_3 L^{-1}.

$$Grey_{leaching} = \sum_{i=1}^{n} Feed_{i,r,m} * \left(\frac{\frac{N_{applied\ i,r,m} * \%leaching_{i,r,m}}{0.2259}}{\max\_nitrate_{drinking-water}}\right) \quad \text{Eq.4}$$

Figure 2 describes all parameters and processes covered in the proposed method for estimation of WU at the inventory level (green, blue and grey water; the latter based on NO_3-N) for the cultivation of feedstuffs.

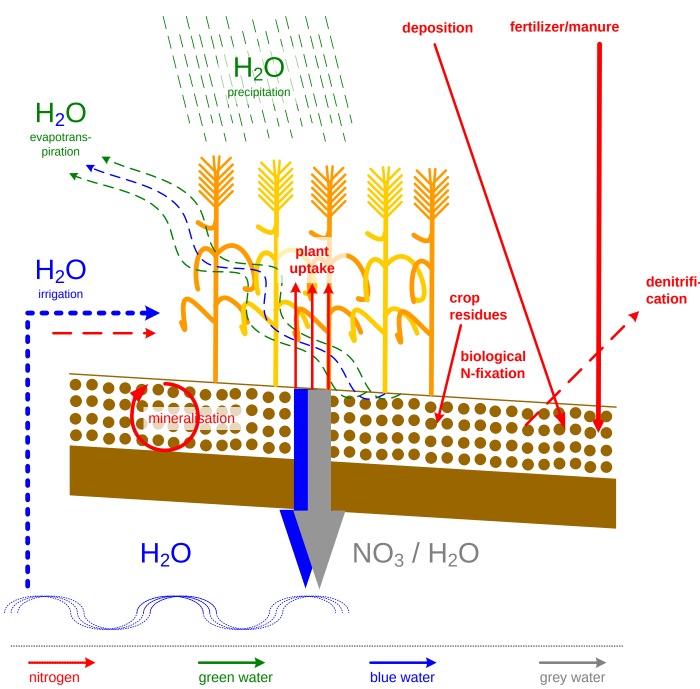

Figure 2: Parameters covered in the proposed method for estimation of water-use concerning cultivation of feedstuffs.

Relatively small amounts of virtual grey dilution water result from potentially leached N related to energy consumption that is primarily emitted as NO_X-N, NH_3-N or N_2O-N from cultivation of feedstuffs, animal husbandry, transports or dairy processing, and is termed $Grey_{energy\text{-}use}$ herein. The amount of leached N is based on the source of energy, i.e. on its specific N emission. According to an IPCC (2006) default value, we assumed 30 % of all gaseous N-emissions from energy supply and usage to be lost to water bodies via deposition on natural land, on settlements without water treatment in sewage plants or on agricultural land, followed by leaching.

As this energy-related grey water is no key source for the overall water demand, a simplified calculation may be used: we converted total energy needed for feed processing, housing, milking, milk-cooling and processes in the dairies, etc. (see Table 1) into diesel-equivalents and calculated the associated CO_2-eq emissions. Based on Hausberger (2000), assumptions concerning NO_X-N, NH_3-N or N_2O-N emissions and the 30 % leached N (IPCC 2006), we calculated 77.3 L of grey dilution water per 1 kg of CO_2-eq from fossil fuels.

The impact-weighted grey water footprint is calculated from a combination of the cultivation-related $Grey_{leaching}$ water demand and the less relevant $Grey_{energy\text{-}use}$ water demand plus a weighting factor to account for a regionally variable "water quality stress".

Two different approaches may be used to derive an impact-weighted grey WF ("Imp.Grey" in equations 5 and 6):

(1) In a regional (catchment based) approach, the ratio between the regional concentration of the substance that defines the grey water (here: nitrate; $\boldsymbol{C}_{regional\ \mathrm{NO3}}$ in mg NO_3 L^{-1}) and its maximum tolerable concentration in drinking water ($\boldsymbol{C}_{max\ \mathrm{NO3}}$, 45 mg L^{-1}) characterize the regional "water quality stress" and is used as an impact factor.

$$Imp.Grey_{leaching-approach(1)} = \left(Grey_{leaching} + Grey_{energy-use}\right) * \left(\frac{\boldsymbol{C}_{\mathrm{regional}\,NO3}}{\boldsymbol{C}_{\mathrm{max}\,NO3}}\right) \qquad \text{Eq.5}$$

(2) For a local approach, the impact-weighted grey WF is derived by using the ratio of grey water needed for dilution of emitted nitrate and the water actually available for dilution as an impact factor. The water available for dilution results from water recharge amounts as a function of local annual precipitation plus irrigation minus evapotranspiration (each per kg of crop).

$$Imp.Grey_{leaching-approach(2)} = \left(Grey_{leaching} + Grey_{energy-use}\right) * \left(\frac{\mathrm{Grey}_{\mathrm{leaching}} + Grey_{energy-use}}{\frac{\left(\sum_{i=1}^{n} Feed_{i,r,m} * (pre_r + I_{i,r,m} - ET_{i,r,m})\right)}{\sum_{i=1}^{n} Feed_{i,r,m}}}\right) \qquad \text{Eq.6}$$

where pre_r is defined as the local precipitation ($Green_{evapotranspiration}$ water plus green water for ground- and surface water recharge) in L per kg harvested yield.

For approach (1) the average impact factors for derivation of the impact-weighted grey WF are 0.336 and 0.334 for the two production systems (PS) "PS A" and "PS L", respectively (see subsection 2.7 for further details on the PS). The associated average values for $C_{regional\ \mathrm{NO3}}$ in drinking water are approximately 10 mg for grassland areas and 25 mg for arable land (BMLFUW/UBA 2006, 2010). For approach (2), we obtain an average impact factor of 1.1928 for PS A with differing recharge water amounts for on-farm areas and land used to produce bought-in concentrates. For PS L we calculated an average impact factor of 1.5268.

2.6. Further sources for blue and red water demand

In addition to blue (and red) water used for irrigation, livestock husbandry requires some blue water consumed by the animals and for cleaning purposes. The drinking water demand of dairy cows and heifers was calculated based on their diets (Wiedner 2010). Average default values from literature (KTBL 2008) were assumed for the required quantity of cleaning and flushing water. Furthermore, some blue water is needed for cleaning purposes in feed processing plants, for cleaning of transport facilities and especially in the dairies

(Nielsen 2003, Theilen and Goldbach 2000). For the red WF, i.e. the impact-weighted blue water demand as described above (subsection 2.1), see the Water Stress Index WSI (Pfister et al. 2009) and subsection 2.3 for further details.

2.7. Input data for Austrian milk production systems and their upstream supply chains

Amounts of feed used to produce one kg of energy-corrected milk were taken from Hörtenhuber et al. (2010) for an alpine and a lowland milk production system (PS “A” and PS “L”). The assessment includes on-farm land and land areas indirectly claimed via bought-in concentrates. Including the effects of the rearing phase and accounting for the by-product beef from cull cows, dairy cows need 0.87 and 0.62 kg DM roughage, mostly from permanent grassland, per kg of energy-corrected milk for PS A and PS L, respectively; an additional 0.16 and 0.18 kg DM of concentrates are fed per kg of milk in PS A and PS L, respectively. The concentrate mixtures are mainly based on barley, wheat and corn as well as some supplementary protein from rapeseed cake, extracted soybean meal and distillers dried grains with solubles; for further details see Hörtenhuber et al. (2010).

Concerning inputs into milk PS as well as their upstream supply chains and calculation procedures described in subsection 2.5, a total of 12.9 L (34 % from fertilizer production) and 13.2 L (20 % from fertilizer production) of $Grey_{energy\text{-}use}$ dilution water demand were calculated for PS A and PS L, respectively. These result from energy consumed for cultivation and processing of feedstuffs, from the production of mineral fertilizers and pesticides, including gaseous N-losses from the fertilizer production (see Table 1), for livestock husbandry and upstream transports.

3. Results

The average sum of unweighted WUs for PS A and PS L is around 700 L per kg milk, with an average grey WU of 314 L/kg (45% of total WUs). In detail, we found 266 and 361 L of grey dilution water per kg milk for PS A and PS L, respectively. Green WU for cultivation of feed was 384 and 304 L/kg of milk for PS A and PS L, respectively; green water lost from the regional cycle due to deforestation added up to 16 (PS A) and 17 (PS L) L/kg. Blue water for irrigation (grains and corn) was identified to be 4 L per kg milk for both PS A and PS L. In total, blue WU estimates for housing, for production of mineral fertilizers and for dairy processing purposes resulted in 13 L/kg (similarly for both PS A and PS L; about each 5 L for housing and the dairy, the other 3 L for production of mineral fertilizers and pesticides). Equally, additional 13 L per kg milk were found for both PS A and PS L for grey water from livestock husbandry’s or transports’ energy demand-related deposited and leached N-emissions. The latter 13 L are accounted for in the category “Grey water for crops and roughage cultivation” in Figure 3, as they are assumed to be deposited mainly on the agricultural land.

The blue and green WU is usually connected to the production efficiency and is therefore higher for the alpine (+25 %) than for the lowland PS, whereas the latter shows greater grey WU (+33 %), especially due to higher nitrate emissions related to the greater proportion of arable land used (see Figure 3). Comparably, the impact-weighted grey WF is also smaller for PS A than for PS L; the difference between the two PS is 32% following approach (1), which reflects a regional perspective, and 72% if estimated by approach (2), which represents the local scale.

In contrast to the impact-weighted grey WF, red water is of very little relevance for Austrian dairy production, because only small amounts of blue water are required, which partially do not leave the local systems and are not scarce in the regions addressed (Pfister et al. 2009). Consequently, red water (less than 1 L per kg milk) is hardly visible in Figure 3.

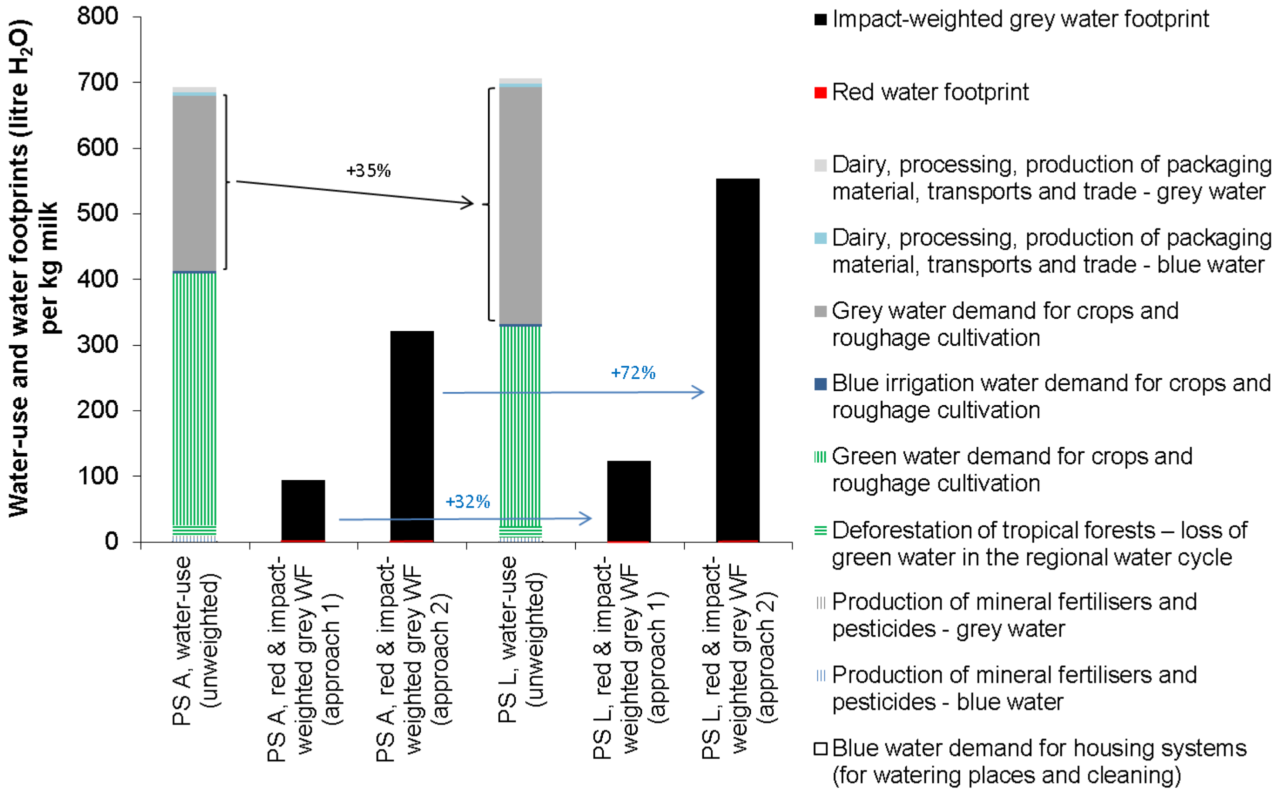

Figure 3: Water-use (blue, green and grey water) and the impact-weighted grey water footprint as well as red water for two dairy production systems in Austria (alpine PS A and lowland PS L).

4. Discussion

Both impact-weighting approaches (1) and (2) seem to be practicable for the impact-weighted grey WFs of typical crops. For specific crops (e.g. with a high proportion of N lost as NO_3) or for a small-structured agricultural area with crop rotations (within one catchment area), we propose to use the less detailed regional approach (1). As a consequence of the less specific influences, approach (1) results in less diverging WF results than approach (2). In approach (2) the impact factor is derived from processes in the upper soil layer of a cultivated area only, but does not account for the dilution effect of water transfer from areas with less or no N fertilization such as forests. Deviating therefrom, the impact factor is derived from a concentration of the relevant substance in affected water sheds in approach (1), which therefore better represents the regional situation, particularly in terms of agricultural impacts (e.g. nitrate losses).

Some conceptual aspects are currently discussed in the relevant literature (see Chenoweth et al. 2013): (i) Should different types of water be added up for a final quantification? While blue and green water represent physically used amounts of water, grey water is a virtual water-use, which is not accompanied by a physical water consumption. Consequently, we did not compare total WU per kg of liquid milk, but compared single sources for water demand. (ii) Green water is naturally supplied by precipitation and thus it is questionable whether it is to be accounted for in a WF, which is assumed to consist only of human-driven water demands in some concepts. Contrarily, we assume green water to have an important impact on available amounts of blue water (in downstream water bodies) and also an impact on grey water, as it dilutes nutrient or pollutant loads. As a consequence, the proposed method accounts for green water at the inventory WU level and estimates an impact-weighted grey WF. This grey WF is influenced by the green water-use, mainly through its function for recharging water bodies. (iii) Most water footprints which implement grey water derive its demand from nitrogen losses, although in many areas other substances may be most limiting for the water quality. Therefore, attention must be given to a proper estimation of the grey water use and to the identification of the substance most critical to water quality, which ultimately defines the grey water demand.

Grey water is not considered in typical multiple-indicator LCA studies, as they usually cover nutrient (nitrogen and phosphorus) loads in an eutrophication potential and partially also pollutant loads (mainly pesticides) in toxicity indicators. To avoid double counting, these studies use at the most blue or red water as an indicator of resource use – if at all. Contrarily, grey water estimates may provide an important information in studies which do not integrate eutrophication potential estimations or toxicity indicators, which not only aim at scientific analysis, but also at the communication of environmental concerns. Regarding the understanding and interpretation of qualitative and quantitative aspects of water resources, the concepts of virtual water and water footprints already provided some knowledge (see Chenoweth et al. 2013). However, in contrast to an eutrophication or a toxicity potential, the impact-weighted grey WF described in this paper even shows an advantage from a scientific perspective: this indicator includes a regionally or locally variable impact of emissions by addressing the background concentration of the substance concerned, such as nitrate, phosphorus or different pesticides.

The method suggested herein basically follows life cycle assessment principles (see guidelines for life cycle assessments: e.g. ISO 2006, BSI 2011) which makes it coherent to other LCA criteria and allows for an integration into a wider assessment, whenever the use of water resources is of interest.

5. Conclusion

We propose an integral consideration of quantitative and qualitative aspects related to water resources in so-called "water footprints", following principles of life cycle assessment. Blue, green and grey water-use shall be assessed at the inventory level and shall be amended by considering red water and an impact-weighted grey water footprint. The latter is defined within the proposed method to characterize the impact of a grey water demand in accordance with an existing method for red water.

For the overall water-use of the two production systems presented herein, an inclusion of grey water leads to a disadvantage for the more intensive PS L as compared to PS A. Similarly, the consideration of the regional and local impact for a WF according to the suggested approaches (1) and (2) leads to a disadvantage for the more intensive PS L due to a substantially higher virtual impact-weighted grey WF.

However, for crops and crop rotations with an inherent risk for water pollution we propose to use the less detailed impact-weighting approach (1), which should be based on measured data, focuses on the regional level and better reflects conditions for small-structured agriculture and the impact of nitrate leaching on regional water quality.

6. References

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Accounting for greenhouse gas emissions from direct and indirect land use change effects: a global approach and case studies

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ABSTRACT

Emissions of CO_2 due to land use and land-use change (LULUC) are being quantified, but their allocation to products is complicated by indirect (iLUC) effects. Arguing that it is increases in net agricultural commodity imports that drive iLUC, we propose to link LULUC emissions from domestic agricultural production to iLUC emissions caused by international agricultural commodity trade. LULUC due to expanding infrastructure areas is discounted. The method is demonstrated with results for selected example nations (e.g. Brazil, China, Indonesia, USA). The method also permits allocating emissions specifically to product groups within the country of consumption. This is demonstrated for product groups within the example countries. Results are found to vary widely between nations and between product groups within a nation. The method is an attempt to provide a computational basis for a more equitable approach to LUC-related GHG-accounting between "LULUC-emitting/exporting nations" and "LULUC-importing nations".

Keywords: land use change, greenhouse gas emissions, LULUC, iLUC

1. Introduction

Increasing demand for food energy and protein adds to the environmental impacts of food and feed production, the latter indirectly through animal production. Impacts include both land use (LU) emissions on extant agricultural areas through intensification and land use change (LUC) related emissions from newly converted areas (such as primary and secondary forests and savannahs; Hörtenhuber et al. 2013, Fargione et al. 2008). LUC and to a lesser extent LU are among the major contributors to global CO_2 emissions, especially in the tropical regions of South-America, Asia and Africa. Emissions from LUC are reported to have contributed approximately 20% of total global CO_2-emissions during the last two decades of the twentieth century (Denman et al. 2007). For the decade from 2000 to 2010 the proportion of CO_2 emissions originating from LUC substantially decreased but still contributed to about 12% (Poruschi et al. 2010) or 10% (Harris et al. 2012) of global CO_2 emissions. The multitude of unanswered scientific questions regarding the complex interactions between LUC's environmental, socio-economic and societal aspects (e.g. in Foley et al. 2005, UNEP 2014) cannot be properly addressed in this paper. This paper focuses on how LULUC-related greenhouse gas emissions can be allocated to the (worldwide) trade and national "consumption" of agricultural commodities.

A previously published method by the authors (Hörtenhuber et al. 2013) attempted to quantify primarily dLUC greenhouse gas (GHG) emissions from known and definable regions of origin. In contrast to the environmental effects of direct land-use-change (dLUC), GHG emissions from indirect land-use-change (iLUC) effects are an evolving issue that has wide-ranging policy implications (e.g. Finkbeiner 2013, Fritsche et al. 2010). Although region-specific dLUC emission results can be useful, they fail to account for the effects of international agricultural commodity trading. Many countries that are global top exporters of food and feedstuffs actually cause domestic LULUC emissions on behalf of the countries buying their exports. We hypothesize that countries with increasing net agricultural exports will tend to emit more CO_2 from LULUC as well, because they are forced to increase production through conversion of previously unused land (LUC) and intensification of cultivation on existing land (causing LU emissions due to soil carbon losses). This potential correlation is of course subject to other factors – for example, a growing domestic population will exacerbate LULUC emissions, while any decrease in domestic demand will weaken export effects.

Consequently, the objective of this work is to provide a deterministic top-down method which accounts for the effects of iLUC linked to international agricultural commodity trade on country-specific LULUC emissions.

2. Methods

In this section we present in more detail the concept underlying the proposed method, some of its key assumptions, and the simple computational steps involved.

2.1. Country-specific shares of agriculture-related LULUC emissions

When accounting for environmental impacts of LU and LUC, agricultural products with increasing production volume exert stronger pressure on land supplies than products with decreasing production volumes. Therefore increasing production should be assigned a larger share of impacts. Conceptually, agricultural exports can be linked to international indirect LUC (iLUC) effects: If domestic production becomes more export-oriented, domestic supply will decrease and the unmet domestic demand will lead to increased commodity imports if economically feasible. The approach presented here thus assumes the existence of a (hypothetical) global pool for iLUC emissions.

Aside from the global iLUC emissions pool, the method presented takes a country-specific approach, since trends in agricultural production, imports and exports differ by region (as well as by product type). A country-specific method can be expected to better mirror large regional LULUC variations than a one-size-fits-all approach that assigns equal LULUC emissions on an area basis (see e.g. Williams et al. 2010, Vellinga et al. 2013), regardless of regional differences. If regional LULUC data and regional agricultural statistics are available within a country, the approach could easily be adapted to a higher spatial resolution as well.

Countries with increasing agricultural exports will feed a proportional share of their LULUC emissions into this pool, thereby reducing their burden of LULUC emissions, and countries with increasing net imports will import a proportional share of the global iLUC pool emissions. It is important to note that this takes a dynamic rather than static view: the temporal changes in exports and imports (increases and decreases) determine the flows of iLUC emissions, and not the absolute export and import data. Also, since the iLUC pool is fed by "exported" LULUC emissions, it actually includes LU-related emissions in addition to LUC-related emissions. Thus the effects of intensified agricultural production on LU emissions are also exported and imported into and out of the pool. We have nevertheless chosen to use the term "iLUC pool" here since it is well established in the international discussion.

To allow for aggregation of the wide variety of agricultural commodities produced by a given country and traded internationally, we convert commodity masses as given in the FAO statistics (FAO 2012) to their energy-content equivalent (lower heating value, LHV), based on data from Fehrenbach et al. (2008), and Beilicke (2010). Furthermore, all calculations in this study refer only to CO_2 emissions; other GHGs such as methane and nitrous oxide are not included in this method, since they typically contribute little to LULUC emissions change (Olivier and Janssens-Maenhout 2011).

As a starting point we use for each country the CO_2 emissions L_k (Olivier and Janssens-Maenhout 2011) that are caused by LULUC:

$$\sum_k L_k = L_{glo} \qquad \text{Eq. 1}$$

where
L_{glo}…annual LULUC emissions, worldwide (excluding those countries for which no suitable data are available), Tg a^{-1}
L_k… annual LULUC emissions from country k, Tg a^{-1}

Each country's LULUC emissions have to be reduced by those LULUC emissions that are caused by the expansion of infrastructure areas (including built-up areas; Krausmann et al. 2013) rather than by the expansion of agricultural areas (FAO 2013). Given the focus on agricultural production, we split the infrastructure LULUC emissions and the agricultural LULUC emissions in proportion to their country-wide area increases.

$$AG_k = L_k - INF_k \qquad \text{Eq. 2}$$

where
AG_k … annual agriculturally-related LULUC CO_2 emissions from country k, Tg a^{-1}

INF_k … annual infrastructure-related LULUC CO_2 emissions from country k, Tg a^{-1}

In the model presented here, agriculture-related LULUC emissions are in principle allocated to the emitting country. As described above, a correction is made to account for iLUC-causing increases of net imports into a country:

$$NL_k = AG_k + NI_k \qquad \text{Eq. 3}$$

where
NL_k...net annual agriculturally-related LULUC emissions for country k, Tg a^{-1}
NI_k...net import emissions due to net import increases into country k, Tg a^{-1}

The following equations illustrate how the net import emissions are calculated.

We first calculate the global iLUC pool and then distribute the iLUC pool's emissions to countries proportional to their net import increases during a selected accounting period.
The global iLUC pool is fed into by all export-increase related LULUC emissions EX_k:

$$iLUC_{glo} = \sum_k EX_k \qquad \text{Eq. 4}$$

EX_k in turn are defined as the share of a country's agriculture-LULUC emissions that is proportional to a country's export increases:

$$EX_k = AG_k * e_k \qquad \text{Eq. 5}$$

where
EX_k...LULUC emissions due to agricultural commodity export increases of country k, Tg a^{-1}
e_k…energy-content (LHV) based export-increase allocation factor (dimensionless).

The allocation factor e_k relates a country's agricultural export increases to its domestic agricultural production increase, both calculated as annual averages and expressed in energy equivalents (LHV):

$$e_k = \frac{\Delta E_k}{\Delta D_k} \qquad \text{Eq. 6}$$

where
ΔE_k…average annual export increases of all agricultural commodities expressed as LHV, TJ a^{-1}
ΔD_k…average annual domestic production of all agricultural commodities expressed as LHV, TJ a^{-1}

The average annual export increase ΔE_k and the average annual domestic production increase ΔD_k are both calculated in the same way: The annual LHV equivalents of exports and production of all products for a given country are averaged over an initial three-year period and over a final three-year period – from 1998 to 2000 and from 2007 to 2009, respectively – and then the initial three-year average is subtracted from the final three-year average to obtain the increase over the ten years of the accounting period (from 1999 to 2008). This ten-year increase is then divided by ten to obtain an average annual increase over the ten years of the accounting period (from 1999 to 2008).

Now that the global iLUC emissions pool has been established, its emissions are distributed among all countries in proportion to their individual net import increases ni_k:

$$NI_k = iLUC_{glo} * ni_k \qquad \text{Eq. 7}$$

where
ni_k…net import-increase allocation factor (LHV-based and dimensionless)

The net import allocation factor ni_k is defined as the difference between a country's share of global import increases and a country's share of global export increases:

$$ni_k = \frac{\Delta I_k}{\sum_k \Delta I_k} - \frac{\Delta E_k}{\sum_k \Delta E_k} \qquad \text{Eq. 8}$$

where
ΔI_k…average annual import increase of all agricultural commodities expressed as LHV, TJ a^{-1}
$\Sigma_k \Delta I_k$…global sum of average annual import increases of all agricultural commodities expressed as LHV, TJ a^{-1}
ΔE_k…average annual export increase of all agricultural commodities expressed as LHV, TJ a^{-1}
$\Sigma_k \Delta E_k$…global sum of average annual export increases of all agricultural commodities expressed as LHV, TJ a^{-1}.

2.2. Product group-specific shares of agriculture-related LULUC emissions

Net LULUC Emissions can also be calculated specifically for a product group p that is consumed in a country k. Emissions will vary for a given product group p depending on the producing country. Otherwise the approach follows largely that for countries.

For aggregating the diverse array of agricultural commodities (i.e., imports, exports, domestic production, and domestic demand), we use the energy content of each product group, expressed in units of lower heating value (LHV). The following product groups are considered here: alcoholic beverages, cereals (excluding for beer), fruits (excluding for wine), oil crops, pulses, spices, starchy roots, sugar and sweeteners, sugar crops, tree nuts, vegetable oils, vegetables, animal fats, eggs, meat, milk (excluding butter), offals, stimulants; no data are available in FAO (2012) for the groups "tobacco and rubber" and "miscellaneous".

Each product group in a country is assigned a share of the country-wide agricultural LULUC AG_k in proportion to its LHV-based share of the total agricultural production:

$$AG_{k,p} = AG_k * a_{k,p} \qquad \text{Eq. 9}$$

where
$AG_{k,p}$...LULUC emissions of agricultural product group p in country k, Tg a^{-1}
$a_{k,p}$ …energy (LHV)-based product allocation factor (dimensionless).

The product allocation factor $a_{k,p}$ relates a product group's production increases in country k to that country's total domestic agricultural production increase, both calculated as annual averages and expressed in energy equivalents (LHV):

$$a_{k,p} = \frac{\Delta P_{k,p}}{\sum_p \Delta P_{k,p}} \qquad \text{Eq. 10}$$

where
$\Delta P_{k,p}$…average annual production increase of product group p in country k, expressed as LHV, TJ a^{-1}

As was done with countrywide emissions, product-specific LULUC emissions $AG_{k,p}$ are adjusted with additional iLUC emissions from the global iLUC pool $NI_{k,p}$ in proportion to their net import increases $ni_{k,p}$. The expression for net LULUC emissions is similar to that for country as a whole:

$$NL_{k,p} = AG_{k,p} + NI_{k,p} \qquad \text{Eq. 11}$$

where
$NL_{k,p}$...net annual agriculturally-related LULUC emissions for product group p in country k, Tg a^{-1}
$NI_{k,p}$...net import emissions due to net import increases of product group p into country k, Tg a^{-1}.

The net import emissions for product group p in country k are calculated as:

$$NI_{k,p} = iLUC_{glo} * ni_{k,p} \qquad \text{Eq. 12}$$

where
$ni_{k,p}$…net import-increase allocation factor for product group p in country k (LHV-based and dimensionless)

The net import allocation factor $ni_{k,p}$ is defined as the difference between a country- and product-specific share of global import increases minus the corresponding global export increases:

$$ni_{k,p} = \frac{\Delta I_{k,p}}{\Sigma_k(\Sigma_p \Delta I_{k,p})} - \frac{\Delta E_{k,p}}{\Sigma_k(\Sigma_p \Delta E_{k,p})} \qquad \text{Eq. 13}$$

where
$\Delta I_{k,p}$…average annual import increase of product group p in country k expressed as LHV, TJ a^{-1}
$\Sigma_k(\Sigma_p \Delta I_k)$…global sum of average annual import increases of product group p expressed as LHV, TJ a^{-1}
ΔE_k…average annual export increase of product group p in country k expressed as LHV, TJ a^{-1}
$\Sigma_k(\Sigma_p \Delta E_k)$…global sum of average annual export increases of product group p expressed as LHV, TJ a^{-1}.

As a last optional step of the method, mass-specific net LULUC emissions $nl_{k,p}$ that are associated with domestic consumption of product group p in country k can be calculated from the average consumption:

$$nl_{k,p} = NL_{k,p} / C_{k,p} \qquad \text{Eq. 14}$$

where
$nl_{k,p}$ …average mass-specific net annual LULUC emissions for consumption of product group p in country k, Tg Tg^{-1} (kg kg^{-1})
$C_{k,p}$ …average consumption of product group p in country k, Tg a^{-1}

All values are averaged over the last three years of the accounting period (2007 to 2009).

3. Results

Following the method described in the previous chapter, we calculated net LULUC emissions for those 175 nations, for which sufficient data were reported in FAO (2012) and Olivier and Janssens-Maenhout (2011). Figure 1 describes average annual net agriculturally-related LULUC emissions (NL_k) per ha of agricultural land, which are combined from a country's agricultural LULUC emissions (AG_k) and the balance NI_k of (a) imported (positive) iLUC emissions and (b) exported (negative) dLUC emissions (see Eq. 3 above). The average global iLUC emissions pool was calculated to be 1.2 Pg per year. This is approximately one fourth of all LULUC-CO_2 emissions from the 175 countries included in the study.

In special cases, countries such as Australia and Japan are assigned no net LULUC emissions (value 0; see also Table 2); in these countries, two conditions applied: (i) neither imports nor exports increased, i.e. no national LULUC emissions are exported to the global iLUC pool, nor is iLUC imported from the pool, and (ii) national LULUC emissions are fully attributed to settlement (infrastructure) area expansion while agricultural land areas declined (compare Eq. 2 above). Other net exporting countries such as Argentina or the USA even show (theoretical) negative net LULUC results per ha (Figure 1). This is a consequence of rapidly increasing (LHV-energy) net export volumes and little or no LULUC import increases (resulting in a negative net import increase balance NI_k), combined with low national LULUC emissions (AG_k). The highest average annual net agriculturally-related LULUC emissions in Table 1 were computed for Indonesia. Of the Indonesian LULUC-related CO_2 emissions, 53 % are attributed to peat fires, 20 % to peat drainage/oxidation, 22 % to deforestation and only 5 % to palm oil and timber plantation establishment (Boer et al 2010, Indonesia's Second National Communication to UNFCCC). This illustrates that emissions may stem not only from deforestation, but also from other LULUC effects. Results for the USA and for Brazil demonstrate details about the calculations and results of net LULUC emissions: Agricultural LULUC emissons AG_k of 0 and 993 Tg were calculated per year in the USA and Brazil, respectively. These national LULUC emissions were corrected by -185 and -110 Tg LULUC emission shares from negative – and therefore export-dominated – net import increases of emissions to the global iLUC pool for the USA and Brazil, respectively. Dividing by the domestic agricultural area ($414*10^6$ ha for USA and $276*10^6$ ha for Brazil), we arrived at net LULUC emissions NL_k per average ha of agricultural land of about -0.3 and +3.0 Mg a^{-1} ha^{-1} for USA and Brazil, respectively.

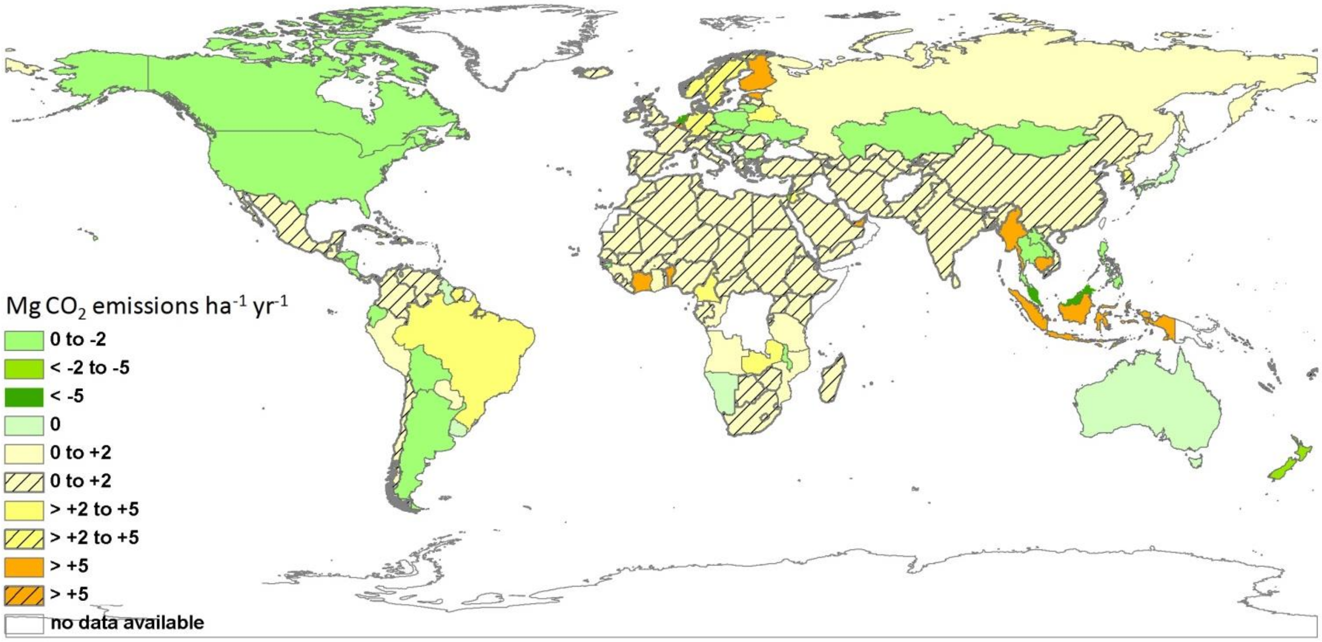

Figure 1. Average annual net agriculturally-related LULUC emissions per ha of agricultural land (Mg ha^{-1} yr^{-1}) corresponding to "NL_k" in Eq. 3 divided by agricultural land area). Hatched areas designate countries where iLUC due to net import increases is more than half of net agricultural LULUC emissions.

In addition to countrywide net agricultural LULUC emissions, we calculated net LULUC emissions specifically for product groups (as listed in FAO 2012) that are consumed within the specific countries. Table 1 illustrates for the example of Brazil the allocation factors ($ni_{k,p}$, Eq. 13) for net-import-increase related iLUC emissions. Some of the allocation factors are negative, indicating net export increases that shift emissions into the global iLUC pool. The product groups with the largest export increases and therefore with the largest negative allocation factors are oil crops (mostly soy), sugars (from sugar cane), and cereals (mostly wheat and maize).

Table 1. Allocation factors for specific product groups' net import-increases for the example of Brazil.

Product group	Allocation factor $ni_{k,p}$ for net import-increase	Product group	Allocation factor $ni_{k,p}$ for net import-increase
Alcoholic Beverages	-1.1%	Sugar crops	0.0%
Cereals - Excluding Beer	-19.6%	Tobacco & Rubber	0.0%
Fruits - Excluding Wine	+0.9%	Tree nuts	-0.1%
Miscellaneous	0.0%	Vegetable Oils	-3.6%
Oil crops	-42.7%	Vegetables	-0.1%
Pulses	0.0%	Animal Fats	-0.2%
Spices	0.0%	Eggs	0.0%
Starchy Roots	+0.1%	Meat	-4.3%
Stimulants	-1.0%	Milk - Excluding Butter	-0.5%
Sugar & Sweeteners	-27.3%	Offals	-0.4%

To complete the picture, the product-specific net LULUC emissions $nl_{k,p}$ are shown in Table 2 for Brazil and other selected countries as well. For Brazil, the strong export growth of oil crops, sugar/sweeteners and cereals (negative contribution to net LULUC emissions) is masked by a larger increase in domestic production $AG_{k,p}$ (Eqs. 9 and 10). However, only for the product group offals are the export increases large enough to result in negative overall LULUC emissions. In contrast, the product groups of tree nuts, vegetable oils, spices and oil crops are assigned large net LULUC emissions, pointing to high domestic emission increases and low export increases.

Table 2. For domestic supplies of individual product groups, average mass-specific net agricultural emissions $nl_{k,p}$ from selected countries in kg CO_2 per kg product (AR = Argentina, AU = Australia, BR = Brazil, CA = Canada, CN = China, FR = France, ID = Indonesia, JP = Japan, UK = United Kingdom, USA = United States of America).

	AR	AU	BR	CA	CN	FR	ID	JP	UK	USA
Alcoholic Beverages	-0.07	0	0.45	0.11	0.00	0.01	0.82	0	0.11	0.03
Cereals - Excluding Beer	-0.71	0	1.70	-0.05	0.03	0.22	4.55	0	0.55	-0.05
Fruits - Excluding Wine	-0.33	0	0.19	0.19	-0.04	0.09	4.07	0	0.34	0.15
Oil Crops	-0.30	0	3.66	-2.49	1.77	0.50	8.70	0	-0.39	-0.95
Pulses	-0.42	0	0.97	-3.41	-0.17	1.76	1.44	0	0.09	-0.7
Spices	-0.02	0	4.61	0.92	-2.87	0.27	8.94	0	1.17	1.13
Starchy Roots	-0.10	0	0.34	-0.32	0.11	-0.05	1.79	0	0.12	0.03
Stimulants	-0.15	0	1.32	1.03	-0.17	0.15	16.16	0	0.57	0.19
Sugar & Sweeteners	-0.67	0	3.17	0.36	0.11	1.03	1.16	0	0.93	0
Sugar Crops	0.02	0	0.67	0.01	0.00	0	0.15	0	0.21	0
Tree Nuts	5.97	0	4.19	1.89	0.18	0.03	28.43	0	2.29	-1.26
Vegetable Oils	-32.36	0	4.88	-3.21	2.64	0.98	131.77	0	1.09	0.95
Vegetables	-0.04	0	0.56	0.03	-0.02	0.09	1.29	0	0.26	0.06
Animal Fats	-4.00	0	1.92	-0.40	-0.07	-0.08	3.64	0	0.04	0.01
Eggs	0	0	0.44	0.04	-0.00	0.07	2.13	0	0.42	-0.12
Meat	-0.19	0	0.72	-0.18	0.01	0.04	3.99	0	0.13	-0.06
Milk - Excluding Butter	-0.10	0	0.35	0.02	0.08	-0.03	12.49	0	0.38	-0.08
Offals	-0.29	0	-0.32	-0.88	1.03	-0.86	2.79	0	0.18	-0.04

The countries of Australia and Japan show no net agricultural LULUC emissions for their product groups – in these countries, agricultural land use decreases, and thus all land expansion is assigned to infrastructure growth according to Eqs. 2 and 9. In addition, both agricultural exports and imports from Australia and Japan decreased during the accounting period. In contrast, export-dominated countries such as Argentina, Canada and the USA show mostly negative net LULUC emissions; in the case of the USA, this applies to fewer product groups than in Argentina. Countries like France and the United Kingdom show positive net agricultural LULUC emissions for most product groups, mainly due to import increases. Emissions for Indonesia are much higher than for the other countries because of large domestic LULUC emissions AG_k, regardless of the product group.

Figures 2 and 3 show for selected countries, which product groups are associated with large positive or negative net LULUC emissions.

Plant-based product groups with high net emissions include spices, stimulants, oil crops, vegetable oils, tree nuts and cereals (Fig. 2). With regard to vegetable oils, Argentina and China are clearly net exporters, and Brazil generally has large positive net LULUC emissions due not to imports, but large domestic production increases. This applies also to production of Argentinean and Brazilian tree nuts.

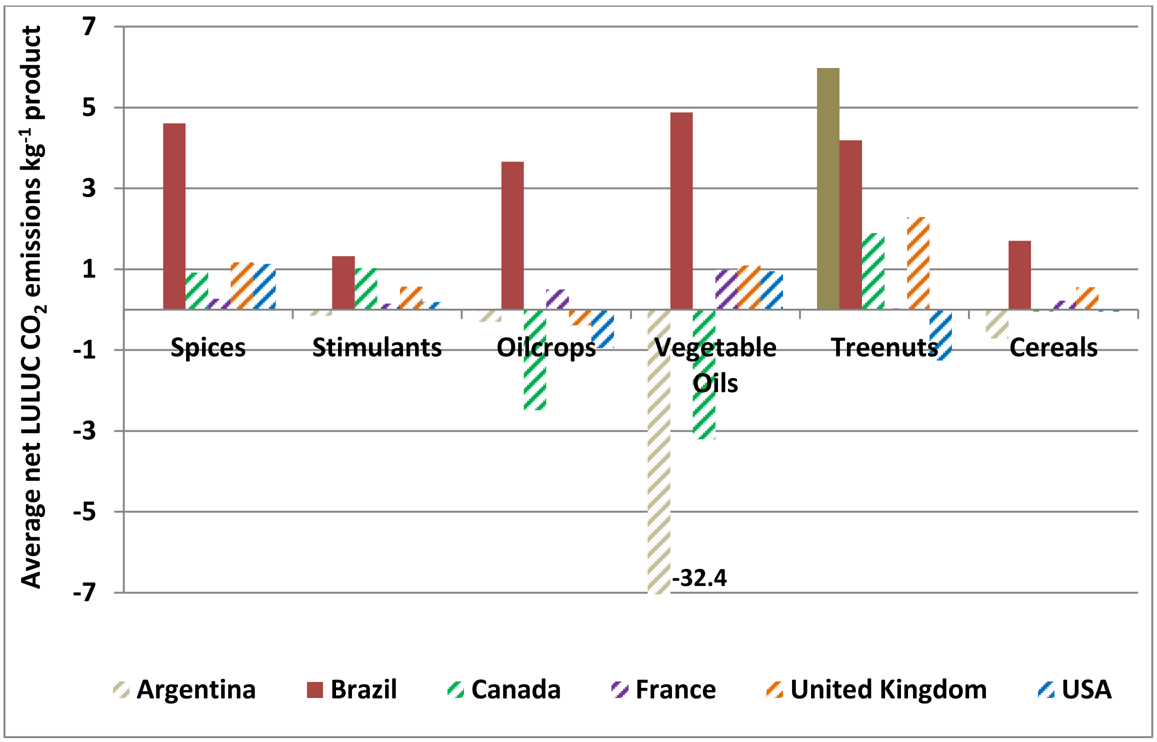

Figure 2. Average net LULUC emissions of specific vegetable product groups with comparably high emissions per kg of product (kg LULUC-CO_2 kg^{-1} product). Hatched columns represent a dominating contribution of iLUC emissions to the net LULUC emissions per unit of product from different groups; solid columns indicate that net LULUC is dominated by emissions assigned to domestic production increases.

Livestock products in Figure 3 show a dominating export role of animal fats and – to a lesser degree – of offal in Argentina, while Chinese imports of offal increased and thus lead to positive net LULUC.

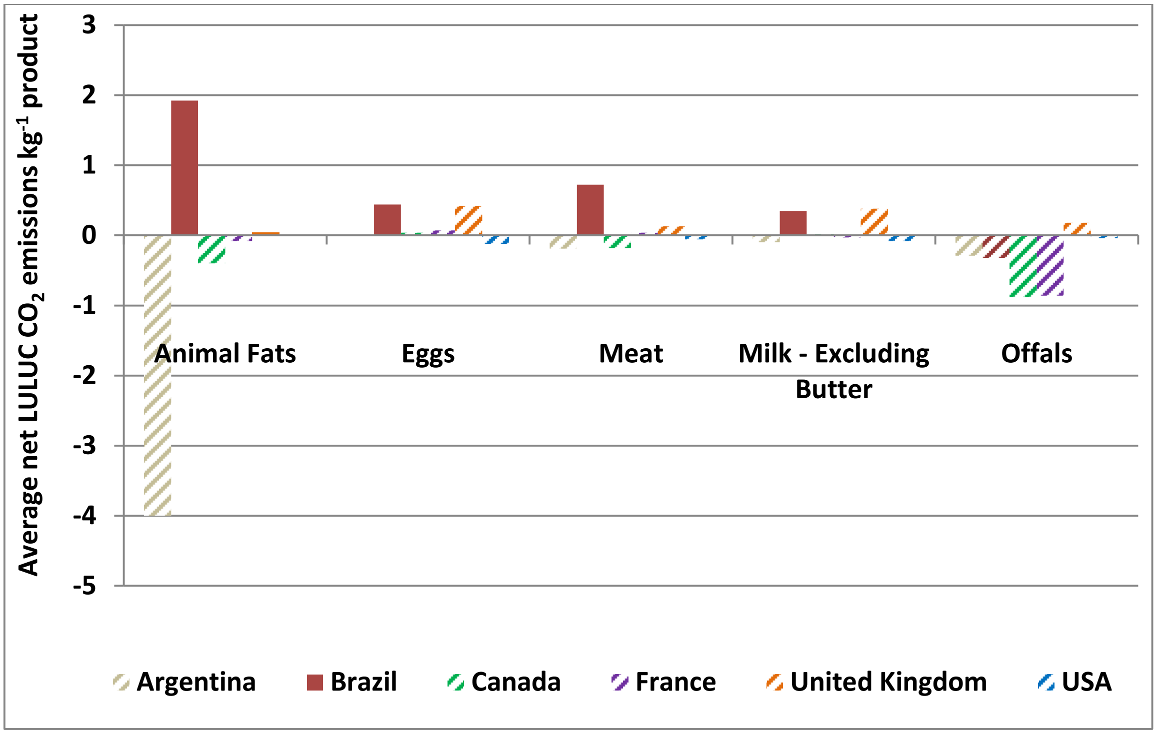

Figure 3. Average net LULUC emissions of specific livestock product groups with mainly comparably high emissions per kg of product (kg LULUC-CO_2 kg^{-1} product).

4. Discussion

Our method assumes that agricultural LULUC is a consequence of increasing demand for agricultural products and thus for land. We derive emission shares based on the dynamic development of agricultural production energy content (e.g., increases of exported energy content), rather than on static, absolute shares of production (e.g. exported energy content itself). This focus on dynamic developments has the advantage of capturing the trends underlying land use change, but it also requires up-to-date information on rapidly changing global agricultural developments, making it hard to extend the method to smaller geographical entities than countries for which such statistics usually are collected (FAO 2012).

By constructing a global iLUC emissions pool, we chose to ignore direct commodity flows between individual countries, to better reflect the global interrelationships of the agricultural commodity marketplace. As stated above, emission shares are allocated in proportion to the energy-content of agricultural product groups. Such an allocation factor has the advantage that it does not fluctuate much over time. As has long been debated in LCA (e.g. Curran 2007) allocation could also be based on commodity prices, but for the purposes of this study the required data were not available. Such an economic aggregation would emphasize the role of monetary drivers for cultivation and agricultural management decisions, but on the other hand, it would be subject to confounding factors such as currency exchange rate fluctuations and fluctuations of auxiliary material prices (fuels, fertilizers, pesticides).

In most countries, the larger part of increased food and feedstuff production is for domestic purposes; in our approach this means that most of a given country's LULUC emissions (globally approximately 70%, authors' calculation) are assigned to that country. The remainder (approximately 30% of worldwide LULUC emissions) are exported or imported and are thus assigned to a global iLUC emissions pool. In many countries though, LULUC from import increases accounts for more than half of the net LULUC (hatched areas in Fig. 1).

In general, country-wide LULUC results suggest that the net LULUC emissions are determined by a combination of a country's agricultural productivity and economic (infrastructure and affluence) development. Product-group specific LULUC emissions vary widely across countries and within a country; high country-wide LULUC emissions – if not caused by infrastructure growth – are a necessary but not sufficient condition for high net LULUC emissions of product groups.

On a general note, the method illustrated here is predicated on the principle of assigning an environmental burden (LULUC emissions) to the consumer (importer of an agricultural commodity) whose demand for the commodity is seen as causing the burden. Conceivably, one could also argue that it is the producer, not the consumer, who decides to satisfy a perceived demand and who therefore should be assigned the LULUC emissions. Applied to LULUC, this shifted perspective would mean that export – related LULUC emissions are still assigned to the producing country/product groups - no "iLUC emissions pool" would be necessary. As a side note, these two opposing approaches are not unknown to LCA practitioners: They are also reflected in two methods used to solve the recycling allocation problem – the cut-off method and the disposal-load method (e.g. Ekvall and Tillman 1997). As a third, compromise approach (similar to the "fifty-fifty" approach in recycling allocation), one could also choose to evenly divide the LULUC emissions from imports and exports between producer and consumer. Mathematically, this would correspond simply to cutting the size of the iLUC emissions pool in half.

Uncertainties of the proposed method's results may be introduced by input data, i.e. FAO (2012, 2013) data concerning areas, yields, national consumption or traded amounts. These data are ultimately collected by the national statistic offices. In addition, the aggregation of single commodities into product groups such as "cereals" causes uncertainties, as different commodities within a group (e.g., types of cereal grains show) will have different lower heating values (LHVs), which even further vary under practical conditions. For example, for the average LHV of the product group "cereals" we used as a surrogate the lower heating value of the globally dominant cereal commodity wheat. A comparison of the wheat LHV with the actual weighed average of the US cereal grain production mix shows a difference of 1.9% between the surrogate and the actual mix. The production mix is based on production data queries (http://quickstats.nass.usda.gov/) of the US Department of Agriculture's statistical data sets for the years 1998-2000 and 2007-2009. Additional uncertainty stems from the conversion of volume based production information (bushels) to mass-based production data, as well as the range of literature-based LHV values for grains. For example, literature values for the LHV of wheat (as fresh matter) range from 14.11 MJ/kg (83 % dry matter) to 14.96 MJ/kg (at 88 % dry matter; Fehrenbach et al. 2008).

5. Conclusion

With the suggested method, we propose an integrated dynamic treatment of LULUC emissions from domestic agricultural production and from iLUC linked to international agricultural commodity trade. We assign LULUC emissions not only to growing agricultural areas in a given country, but also to increases in land occupied by infrastructure and built-up land. Ignoring the latter LULUC emissions (caused by infrastructure growth) and focusing instead on agriculture-related LULUC emissions, we calculate net country-specific agricultural LULUC emissions. We do this essentially by adding LULUC emissions from increased domestic production and iLUC emissions from a country's net import increases. Net import increases are the balance of (a) imported (positive) iLUC emission increases and (b) exported (negative) dLUC emission increases. Using increases of production, exports, or imports in place of their absolute values allows a dynamic rather than static accounting of agricultural commodity production and trade.

Data on agricultural commodity production and trade are converted to the commodity's corresponding energy content as LHV. This allows aggregation of agricultural product group data on a national and global scale.

A country's iLUC emissions are allocated from a (hypothetical) global iLUC pool; this corresponds to assuming a worldwide average import mix which reflects the global interconnectedness of agricultural commodity trade und thus "traded" LULUC emissions.

In addition to national results (e.g. per average hectare of agriculturally used land), our method is able to allocate emissions specifically for product groups that are consumed in a country. The method's allocation of emissions on diverse product groups provides new results with indirect (LU)LUC effects for 175 nations (with full data).

We demonstrate this approach for selected nations (e.g. Brazil, China, Indonesia, USA) by presenting net average annual emissions per ha for each selected country in general and for these countries' product groups in particular (e.g. for cereals, oil crops or livestock products such as animal fat, milk or meat). The application of the method delivers interesting results for the analyzed nations, with widely varying results between nations on one hand and product groups within a nation on the other hand.

Due to the novelty of the method, we have unfortunately not been able to compare our results to those of similar LCA-related dynamic approaches, which account for international agricultural commodity trade. A comparison of our method's results with those of direct (LU)LUC approaches seems not appropriate: By accounting for country-specific domestic LULUC and iLUC net-emissions, we attempt a more spatially accurate attribution of iLUC effects than existing iLUC models. We are thus suggesting a computational basis for a more equitable approach to LUC-related GHG-accounting between "LULUC-emitting/exporting nations" and "LULUC-importing nations" as well as between (LU)LUC-driving product groups and product groups with little or no effects on LULUC-emissions. Further work should address the issue of the verification and improvement of the model and its input data.

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Water availability footprint of dairy production in Northeast China

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ABSTRACT

As China's dairy consumption is growing, both the domestic milk production and the importation of dairy products are increasing to meet demand. Life cycle assessment (LCA) has recently been applied to assess the water availability footprint for dairy production systems in the major production region of Heilongjiang (Huang et al. 2014). Comparisons were also made with milk produced in the US (California) and New Zealand. The water footprint of milk (cradle to farm gate) produced in Heilongjiang was around 11 L H_2Oe (H_2O-equivalents) kg^{-1} fat-protein-corrected milk (FPCM). This compared to 461 and 0.01 L H_2Oe kg^{-1} FPCM for milk in California and New Zealand respectively. These results suggest that large-scale milk production systems in north-east China, with little dependence on irrigation, have only a modest impact on freshwater availability. Further expansion of the Chinese dairy industry should avoid farming systems with high consumptive water requirements in water-stressed regions.

Keywords: dairy farming, agri-food sector, life cycle assessment, water footprint, sustainable water use

1. Introduction

The environmental burdens of livestock production are a major concern (Herrero and Thornton 2013), especially impacts related to greenhouse gas emissions, land and water use (IDF 2010; Mekonnen and Hoekstra 2012; Stehfest et al. 2013). As livestock production in developing countries is expected to increase, these problems are likely to become even more pressing. China is the world's most populous country and with rapid economic growth diets have shifted toward more calories from animal fats and proteins. From 1990 to 2009, China's dairy consumption increased from 6 to 30 kg per capita per annum (FAO 2012). In response to the growing demand, domestic milk output has increased as well as net imports of dairy products. The expanding Chinese dairy industry has been examined from a range of environmental perspectives (Liu et al. 2004; Sun et al. 2010; Wang et al. 2010). This paper, based on a recent article (Huang et al. 2014), specifically addresses concerns related to water depletion, which is a prominent environmental concern in much of northern China.

According to previous studies using the virtual water approach, which only reports the volumes of water used in production, the increase of livestock production was reckoned as the main driver of China's water scarcity (Liu et al. 2008; Liu and Savenije 2008). However, this argument is built on the general situation that animal products have higher virtual water than crop products (e.g. 1,644 m^3 ton^{-1} of cereals; 15,415 m^3 ton^{-1} of beef) (WFN, 2012). Such volumetric water footprints, which give no consideration to the environmental relevance of water being used, have been described as potentially confusing and misleading (Ridoutt et al. 2009; Zonderland-Thomassen and Ledgard 2012), often giving the largest values to rain-fed agricultural production systems which are actually not associated with any water withdrawal. Environmental relevance should be taken into consideration if the water footprint indicator is to inform wise decision making and policy development (Pfister et al. 2011; Ridoutt and Huang 2012), and this is the direction a new international standard for water footprint is taking (ISO 14046). Several LCA-based water footprinting methods which enable accounting and impact assessment of water use are now available (e.g. Bayart et al. 2010; Mila i Canals et al. 2009; Pfister et al. 2009; Ridoutt and Pfister 2010). Compared to volumetric-oriented indicators such as virtual water, impact-oriented indicators are recommended as being more revealing for decision making (Berger and Finkbeiner 2013; Ridoutt and Pfister 2010; Zonderland-Thomassen and Ledgard 2012).

However, to our knowledge, the application of LCA-based water footprinting to Chinese dairy production is scanty. We conducted a detailed inventory of life cycle water consumption of dairy production in Northeast China (Huang et al. 2014). The water footprints of milk of Chinese origin were subsequently compared with milk produced in countries which are significant in their exports of dairy products to China. Our main purpose was to offer strategic insights to the Chinese food industry about ways to improve the sustainability of products

Jing Huang and Ling Du contributed equally to this study.

containing dairy ingredients from a consumptive water use perspective. The wider dissemination of results is also intended to increase understanding of sustainable water use in relation to China's expanding dairy industry.

2. Methods

2.1. System description

The growth of China's milk output over the past decades has mainly resulted from the expansion of the national dairy herd (0.64 million head in 1980 to 14.2 million head in 2010, DAC 2011) and a shift in production patterns from household to large-scale dairy farms. Milk production mainly occurs in northern China, which in 2010 had more than 80% of the national herd (NBSC 2011). This study concerns the large-scale milk production systems in the north-eastern province of Heilongjiang which is the second largest in terms of milk output. In this region, dairy cows are predominantly raised in mixed farming systems where crop products such as maize, wheat bran and soybean meal are used as feed for the cows. The annual rainfall ranges between 400 and 600 mm, and is concentrated during spring and autumn. Supplementary irrigation is necessary for maize, wheat and soybean. Forage crops such as silage maize and grass are generally rain-fed.

2.2. Life cycle inventory

An LCA based-water footprinting method was used to assess consumptive water use in the production of raw milk (Ridoutt and Pfister 2010). This study only took into account the way the production system limits the availability of freshwater for the environment and for other human uses. As such, only the consumptive water use from surface and groundwater (so-called blue water) was considered. The consumption of soil moisture derived from natural rainfall (so-called green water) was included to estimate the blue water consumption. However, green water will be disregarded in the impact assessment phase (section 2.3) because green water does not generally contribute to regional freshwater scarcity in water bodies (Mila i Canals et al. 2009; Ridoutt and Pfister 2010).

The dairy farming subsystem was modeled using first-hand survey data collected from four large-scale dairy farms and covered the 2011 financial year (Table 1). Capital goods used in production (e.g. machinery and buildings) and dairy farm services (e.g. veterinary and business services) were not included in the assessment as these items are difficult to ascertain and are of minor environmental relevance in most open farming systems (Ridoutt et al. 2012). Maize, maize silage and hay were either grown on-farm or purchased from local crop farmers. Soybean meal and wheat bran were purchased from local oil extraction and grain milling industries which were processing local crop products. In this region, on-farm water use was from groundwater. Effluents such as urine and dung were regularly collected and stored in ponds for 30 to 180 days then returned to local croplands for feed production. Further information regarding the irrigation and other farm inputs used to produce purchased feed crops were collected from local farms and by consultation with local experts (Table 2).

Water consumption associated with the production of farm inputs (fuels, fertilizers, etc) was determined using the Chinese Life Cycle Database (CLCD; www.itke.com.cn). Water consumption was expressed relative to the amount of fat and protein corrected milk (FPCM, 4.0 % fat and 3.3 % protein, FAO 2010). For the allocation of water consumption from processes with multiple co-products, there are several available methods such as the biophysical, economic and protein-based allocation methods (Gerber et al. 2010; Guinée et al. 2004; Thoma et al. 2013). This study applied both the biophysical and economic approaches, enabling the results to be compared with other studies applying these same methods. Application of the biophysical approach followed Thoma et al. (2013). The incoming feed energy was estimated from the surveyed quantity of beef and milk produced and the known nutritional characteristics of the specific feeds consumed by the animals (Gao 2009). The ratio of the feed energy deposited as milk to the total feed energy deposited as milk and beef was then used to define the allocation ratio. The economic approach to allocation used average prices over the last five years.

2.3. Impact assessment

Impact assessment was used to assess the environmental relevance of consumptive water flows in relation to freshwater scarcity. Local characterization factors for freshwater consumption were taken from the Water Stress

Index (WSI) of Pfister et al. (2009). The WSI of the study region was 0.125 and the national average WSI for China (0.478) was used in relation to farm and industrial inputs where the location of production was uncertain. As for the water footprint, each instance of consumptive water use was multiplied by the relevant WSI and then summed across the product life cycle. To enable comparisons between products produced in different regions, the water footprint was then normalized by dividing by the global average WSI (0.602; Pfister et al. 2009) and expressed in the units H_2O equivalents (H_2Oe).

Table 1. Characteristics of the dairy farming subsystems in Heilongjiang, China.

Variable [a]	**Value**
Livestock	
Average number of heifers <2 yr old, head	90
Average number of milkers, head	110
Average number of dry cows, head	30
Average number of bulls, head [b]	0
Average number of mortality and replacement, head	15
Annual milk production, t farm^{-1}	693
Fat content, %	3.5
Protein content, %	3.0
Feed	
Maize, t farm^{-1}	439
Maize silage, t farm^{-1}	1128
Soybean meal, t farm^{-1}	165
Wheat bran, t farm^{-1}	97
Hay, t farm^{-1}	222
Other Farm inputs	
Drinking water use, t farm^{-1}	7300
Dairy shed water use, t farm^{-1}	1460
Electricity, kwh farm^{-1}	2000
Coal, t farm^{-1}	20
Diesel, t farm^{-1}	4

[a] All figures are presented on a yearly basis, yr^{-1}.
[b] Artificial insemination

Table 2. Characteristics of the crop farming subsystems in Heilongjiang, China.

	Maize (grain)	**Maize (silage)**	**Wheat**	**Soybean**	**Hay**
Irrigation, m^3 ha^{-1}	266	0	1260	115	0
Diesel, kg ha^{-1}	23	0	22	21	0
Pesticide, kg ha^{-1}	1.9	2.4	1.3	2.8	0
Fertilizer, kg ha^{-1}					
N	160	200	240	45	60
P_2O_5	50	90	130	80	19
K_2O	45	45	53	45	50
Yield, kg ha^{-1}	9500	85000	3350	1850	2000

2.4. Sensitivity analysis

LCA studies are frequently influenced by uncertainties arising from different factors (Bjorklund 2002). Following ISO 14044 (2006), this study applied sensitivity analysis to the allocation rules and data uncertainty, using the data collected from high and low input farms identified from the four that were surveyed.

2.5. Comparison of milk produced in other countries

The water footprint of milk produced in Heilongjiang was compared with production in the US (California) and New Zealand as imports of dairy ingredients from these regions are important to the Chinese food industry. Lacking detailed information on dairy farms in these two regions, related data for raw milk at farm gate were collected from literature and adjusted to the FPCM functional unit where necessary (Asselin 2012, USDA 2007, Zonderland-Thomassen and Ledgard 2012).

3. Results

3.1. Water consumption of milk produced in Heilongjiang and sensitivity analysis

For large-scale dairy production systems in Northeast China, the allocation of resource use to milk was 92.3% using a biophysical allocation method and slightly higher at 95.2% using an economic allocation method. Consumptive water use also varied between farms, such that the water use allocated to the production of 1 kg fat-protein-corrected milk (at farm gate) varied from 65.4 to 75.3 L (biophysical allocation method) and from 67.3 to 77.7 L (economic allocation method) (Table 3). The average blue water consumption across the four surveyed dairy farms in Heilongjiang was 69.0 L kg^{-1} FPCM (biophysical allocation), with 83% occurring in the production of feed (Table 4). The sensitivity analysis indicated that the variations between high and low input farms were more important than choice of allocation method. That said, the uncertainty associated with data was not regarded as significant in terms of influencing the comparative ranking of water footprint results for milk originating from Heilongjiang, NZ and California (Section 3.2 following).

Table 3. Water consumption of milk produced in Heilongjiang and sensitivity check on allocation rule.

Water consumption per unit milk	High input farm	Low input farm	Difference
Biophysical allocation method, L kg^{-1} FPCM	75.3	65.4	9.9
Economic allocation method, L kg^{-1} FPCM	77.7	67.3	10.4
Deviation, L kg^{-1} FPCM	2.4	1.9	0.5
Deviation, %	3.2	2.9	5.1

Table 4. Water consumption of milk produced in Heilongjiang and sensitivity check on data uncertainty.

Water consumption per unit milk [a]	Feed production	Dairy farm operation	Total
Average value, L kg^{-1} FPCM	57.3	11.7	69.0
High input farm, L kg^{-1} FPCM	62.8	12.5	75.3
Deviation, L kg^{-1} FPCM	5.5	0.8	6.3
Deviation, %	9.6	6.8	9.1
Low input farm, L kg^{-1} FPCM	54.7	10.7	65.4
Deviation, L kg^{-1} FPCM	-2.6	-1.0	-3.6
Deviation, %	-4.5	-8.5	-5.2

[a] The calculation was based on the biophysical allocation method.

3.2. Water footprint of milk produced in Heilongjiang, California and New Zealand

The average water footprint of milk produced in Heilongjiang was 11 L H_2Oe kg^{-1} FPCM (by biophysical allocation), with 76% occurring in the production of feed (Table 5). This can be interpreted as meaning that the production of 1 kg milk in Heilongjiang had an equivalent potential to contribute to freshwater scarcity as the direct consumption of 11 L of water at the global average WSI. Milk produced in California had a higher water footprint (461 L H_2Oe kg^{-1} FPCM), with relatively high consumptive water use (470 L kg^{-1} FPCM) and high water scarcity. New Zealand milk had the lowest water footprint (0.01 L H_2Oe kg^{-1} FPCM, Table 5).

Table 5. Water footprint of milk produced in Heilongjiang, California and New Zealand at dairy factory gate.

	Heilongjiang, China	California, USA	New Zealand
Blue water consumption [a], L kg^{-1} FPCM	69	470	1
WSI (local) [b]	0.125	0.996	0.011
WSI (national) [b]	0.478	0.499	0.023
Water footprint [a], L H_2Oe kg^{-1} FPCM	11	461	0.01
Feed production, %	76	>95	<5
Dairy farm operation, %	24	<5	>95

[a] Based on data from Asselin (2012) and Zonderland-Thomassen and Ledgard (2012).
[b] Data from Pfister et al. (2009).

4. Discussion

4.1. Water footprint of large-scale dairy farming in China

For large-scale milk production systems in Heilongjiang province, most consumptive water use is associated with feed production and this life cycle stage is the first focus for mitigation. Strategies for reducing consumptive water use might include increasing irrigation water use efficiency in crop production, importing feed from other regions where less irrigation is needed or where local water scarcity is even lower, and increasing feed conversion efficiencies for milk production. Although water consumption associated with other dairy farm activities (e.g. animal drinking water, dairy shed water use and water associated with farm inputs) was small, strategies such as reducing the evaporation and wastage of drinking water and the use of recycled water for cleaning could have additional potential to reduce water consumption.

To appropriately address the environmental impacts of consumptive water use, the local water scarcity where production occurs must be taken into consideration. The water availability footprint of milk produced in Heilongjiang was only 11 L H_2Oe kg^{-1} FPCM, much lower than that of cereals produced in some of China's water-scarce regions (e.g. 367 L H_2Oe kg^{-1} for maize in Huang basin, 931 L H_2Oe kg^{-1} for wheat in Hai basin) (Huang et al. 2012). Although plant and animal-derived products cannot be directly compared because of differing nutritional attributes, these water footprint results illustrate that animal products can have lower environmental impact in terms of water resource depletion than some crop products. Even for the same agricultural commodity, farming systems and local environmental context can differ greatly. When considering a large country, such as China, variation in local water stress can be extreme with WSI values (at the province scale) ranging from 0.02 to 1.00 (calculated using data from Pfister et al. (2009)).

The blue water consumption associated with milk produced in grazing and mixed systems has typically been found to be less than that in industrial dairy systems where all feed components are purchased and there is a greater tendency to rely on irrigated crops (Mekonnen and Hoekstra 2012). In China, the majority of dairy cows are raised in these grazing and mixed farming systems (DAC 2011). In addition, the relatively low water stress in Heilongjiang resulted in the milk products derived from this region having only a modest water footprint. As a general principle, low resource input, predominantly non-irrigated, grass and crop-based livestock production systems have little impact on freshwater resources from consumptive water use. As such, similarly modest water footprint results might be expected for livestock products in many other parts of China and the general assertion that livestock production is a major driver of water scarcity in China should not be hastily accepted. We recommend that the water footprints of dairy and other livestock products produced in the different regions of China should be explored in future research using the LCA approach.

To meet the increasing demand for dairy products, China needs to significantly expand its dairy production. As mentioned in section 2.1, more than 80% of the milk production currently occurs in northern China. Unlike Heilongjiang, some of the other provinces in the north (e.g., Inner Mongolia, Hebei and Shanxi provinces) have regions which are already under serious water stress. Increased water consumption for milk production in these provinces may cause serious environmental impact. Water footprint assessment is therefore essential to guide the sustainable development of China's dairy industry.

4.2. Impact mitigation options for the agri-food industry

Nowadays, many consumers have become more critical and not only wish to be informed about the safety of their food but also its origin and the sustainability of its production (Wognum et al. 2011). Manufacturers in the food sector can respond to these changing consumer demands by implementing strategies to reduce environmental impact and by public reporting of product environmental footprints. This study has highlighted a range of practical interventions for food companies to reduce the negative impacts arising from water consumption in the life cycles of food products with dairy ingredients.

Due to the importance of water consumed in primary production, one approach that food companies can take is to source agricultural ingredients from locations with low water stress and from regions where there is low irrigation water demand. For example, Page et al. (2011) reported the water footprint of tomato grown for the Sydney market in Australia and found that the water footprint of local produce was about 8 times higher than alternative suppliers from other regions. Similarly, the water footprints of beef ranged from 3 to 221 L H_2Oe kg^{-1} at the farm gate between six beef cattle production systems in Australia (Ridoutt et al. 2012). These results illustrate the enormous variability in water footprints that exists and therefore the great potential to alleviate water scarcity through selective procurement of agricultural commodities. In the case of dairy-based confectionary (Huang et al. 2014), dairy ingredients supplied from California, where both the local WSI and irrigation water demands were highest, might be avoided or otherwise highlighted for strategic water footprint reduction initiatives. Another consideration to reduce the water footprint of the dairy-based processed foods might be to source more dairy ingredients from New Zealand or Heilongjiang. That said, for some ingredients, it may be more difficult to change the supply region in the short-term because of the structure and location of the established agricultural industries. The opportunities for impact mitigation in such cases are to source ingredients from farms which have higher irrigation water use efficiency within the region. Investments in farming systems which increase the efficiency of irrigation water use, decrease runoff and increase the productivity of rain-fed production systems should be encouraged. These activities could be driven by the food industry by making water-saving agreements with their suppliers.

4.3. Limitations of this study

This study has been the first application of LCA-based water footprinting in Chinese dairy production. There are several limitations arising from the available data and calculation method. Firstly, the dairy farming subsystem was modeled using data collected from only four dairy farms. Although the sensitivity analysis indicated that the overall findings were not affected by farming system data uncertainty, it remains desirable to expand the sample size in future research and to include additional regions of milk production. Furthermore, this study only assessed consumptive water use in the context of water stress. However, water pollution is also an acknowledged problem in parts of China and in certain cases the dairy sector is implicated (Liu et al. 2004; Wang et al. 2010). It is therefore desirable that future water footprint research of dairy products in China investigate nutrient and other emissions to water in addition to consumptive water use.

The limitation of the water footprint metric is that it focuses on the single issue of water scarcity and is not an indicator of overall environmental sustainability. There are frequent tradeoffs between water consumption, greenhouse gas (GHG) emissions and other sources of environmental impact. For example, tomato production systems in Australia with lower water footprint were also found to have a higher carbon footprint (Page et al. 2012). Actions to reduce water footprints might require more energy use and consequently increase GHG emissions, and interventions to reduce GHG emissions might increase water consumption. Therefore, priorities for overall environmental improvement should be carefully considered.

5. Conclusion

As the first application of LCA-based water footprinting in the dairy industry in China, this study has illustrated that livestock products can be produced with modest potential to contribute to freshwater scarcity. Thus, the generalization that the growing demand for livestock products is the major driving factor for China’s water scarcity is not supported in this case. We conclude that it is necessary to examine the regionalized variation in water footprints of all major agricultural commodities as the heterogeneity within sectors is large and the opportunities for water footprint reduction are widespread. This study has demonstrated the large variability in the water footprints between dairy farming systems. As China's domestic milk production is expanding to meet the growing demand, expansion of dairy farming in water-stressed regions should be avoided, unless dairy systems in these areas can predominantly rely on rain-fed crops and pastures. Strategic opportunities also exist for China to reduce its external water footprint associated with dairy products imported from other countries. Food companies can reduce their burden on freshwater systems by sourcing dairy ingredients from regions with low water stress and low water consumption demand. However, interventions to reduce water footprints should not be taken without due consideration given to the potential consequences for other environmental impact categories (e.g. GHG emissions), as well as social and economic factors.

6. Acknowledgement

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Analysis of the determinants of the economic and environmental performance of Swiss dairy farms in the alpine area

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ABSTRACT

Improving the sustainability of the dairy food chain involves a reduction of the environmental impact of dairy farming, as a large part of the environmental impacts associated with dairy product consumption is generated in the agricultural phase of the milk life cycle. In this paper, we combine life cycle assessment and farm accountancy data to analyze the factors affecting the environmental performance, defined as the eco-efficiency of food production, and the economic performance of Swiss dairy farms in the alpine area. The results of the analysis show the existence of synergies in the enhancement of farm economic and environmental performance. Unfavorable natural production conditions (high altitudes and unfavorable topography) have a strong negative impact on both areas of performance. Conversely, organic farming, farm size, full-time farming and a high agricultural education level have a positive effect on the two dimensions of the sustainable performance of a farm.

Keywords: sustainability, dairy farming, economic performance, environmental performance, Switzerland

1. Introduction

Dairy products are of high relevance in terms of environmental sustainability of final consumption. According to a study conducted for the EU-25 by Tukker et al. (2006), dairy products are—within the food and drink consumption area—the second highest contributors[1] to the environmental impact of final consumption by private households and the public sector. Only a few studies have assessed the relative contribution of each phase in the life cycle of milk to milk's total environmental impact over its whole life cycle from production through consumption to disposal. Focusing on the milk production and processing phases, Hospido et al. (2003) showed for the Galician dairy sector that of these two phases the production phase (farming) was—for the impact categories (i) global warming potential, (ii) eutrophication potential and (iii) acidification potential—the main contributor to the total environmental impact generation (with a contribution to the total impacts of 80%, 74% and 58%, respectively). Performing a comprehensive life cycle assessment encompassing the farming, processing and consumption phases, Eide (2002) showed for Norwegian dairies that the agricultural phase was—for (i) the energy consumption, (ii) the acidification potential, (iii) the eutrophication potential and (iv) the global warming potential—the greatest contributor to the total environmental impact of the whole dairy supply chain. Assessing a very large sample of dairy farm operators from the United States and considering all phases in the dairy supply chain, Thoma et al. (2013) found that 72% of the greenhouse gas emissions associated with the consumption of fluid milk in the United States was accrued by the dairy farm gate. Analyzing—within a comparative study between Switzerland, Germany, France and Italy—the life cycle of cheese up to its point of sale, Bystricky et al (2014) found that the farming stage was responsible—for all environmental impact categories considered (demand for non-renewable energy resources, global warming potential, ozone formation potential, land use, eutrophication potential, acidification potential, terrestrial and aquatic ecotoxicity, and human toxicity) for more than 70% of the environmental impacts generated from the "cradle to the point of sale". These four studies provide evidence that, within the dairy supply chain, the "cradle-to-farm gate" link is for most environmental impact categories the most important contributor to the environmental impact of the full chain. A thorough understanding of the factors affecting the environmental impact generation at this level is therefore a pre-requisite if we want to improve the environmental sustainability of the dairy food chain and thus reduce its contribution to the environmental impacts related to the final consumption of products by private households and the public sector.

The present paper focuses on the Swiss dairy food chain of the alpine area, which is of particular importance for the Swiss agricultural sector (see Jan et al. 2012b). By using life cycle assessment (LCA) in combination

[1] The most important contributor is meat and meat products.

with farm accountancy data, we aim to identify the factors influencing the environmental and economic performance of Swiss dairy farms located in the hill and mountain region.

2. Materials and methods

The present work is based on the same data as those used in Jan et al. (2012a). Hence, we forgo a comprehensive description of this dataset and refer the reader to this publication for detailed information on the data and especially on the environmental impact assessment carried out.

2.1. Sample of farms

The investigation relied on a pooled[2] sample of specialized dairy farms located in the hill and mountain regions. The sample encompassed 56 farm observations. The hill and mountain regions included the hill zone as well as the mountain zones 1 to 4 as defined in FOAG (2008). The hill and mountain regions, also called alpine area in the present paper, can be defined roughly as the agricultural production area located between 500 and 1'500 meters above sea level. A specialized dairy farm was defined as a farm whose revenues from dairying generated at least 60% of total farm agricultural revenues without any direct payments. Farms with a proportion of revenues from para-agricultural activities above 20% of total farm revenues as well as farms whose revenues from forestry activities generated more than 10% of total farm agricultural revenues were excluded from the analysis to ensure that the observations were homogeneous in terms of production activities.

The data were collected within the framework of a broader project, the LCA-FADN (Life Cycle Assessment–Farm Accountancy Data Network) project, aiming at conducting a joint economic and environmental assessment of Swiss agriculture at the farm level (see Hersener et al. 2011). The farms of the sample were not selected according to a random procedure. The participation in the project occurred on a voluntary basis due to the complexity and comprehensiveness of the environmental data collection.

2.2. Environmental impact assessment using the SALCA approach

For each farm, a precise and comprehensive environmental impact assessment was conducted by using the SALCA (Swiss Agricultural Life Cycle Assessment) approach (see Baumgartner et al. 2011). The system investigated was made up of the agricultural production system defined in a narrow sense (see Jan et al. 2012a). The assessment covered the agricultural stage, that is, the "cradle-to-farm gate" link, of the milk life cycle. All agricultural inputs, production processes and outputs were taken into account. The environmental impacts were quantified based on very precise and detailed production inventories collected at the farm level. The following eight impact categories were quantified (the impact assessment method used being given in parentheses): (i) demand for non-renewable energy resources (ecoinvent method, Frischknecht et al. 2004), (ii) global warming potential over 100 years (IPCC method, IPCC 2007), (iii) eutrophication potential (EDIP97 method, Hauschild and Wenzel 1998), (iv) acidification potential (EDIP97 method, Hauschild and Wenzel 1998), (v) aquatic ecotoxicity (CML01 method, Guinée et al. 2001), (vi) terrestrial ecotoxicity (CML01 method, Guinée et al. 2001), (vii) human toxicity (CML01 method, Guinée et al. 2001) and (viii) land use (CML01 method, Guinée et al. 2001).

2.3. Analysis level

In the present work, the analysis was carried out at the whole-farm level and not at the level of the product "milk." This choice was motivated by the high degree of specialization in dairying of the farms investigated and by the associated homogeneity of their product mix (i.e., of their production activities). Choosing the farm unit instead of the product group "milk" as an analysis level furthermore enabled us to circumvent the typical problem of allocation of resources and emissions to different products (or product groups) encountered in life cycle assessment at product level in a multiple-product setting (e.g., Feitz et al. 2007, who showed—using a dairy manufacturing plant as a case study—how sensitive the product-level results of LCA studies were to the allocation approach used).

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[2] The observations of a three-year period from 2006 to 2008 were pooled.

2.4. Economic performance

There exist many possible indicators to assess the economic performance of a farm. Basically, these indicators can be divided into two sub-groups: (i) the efficiency measures from the field of productive efficiency measurement and (ii) the classical profitability indicators commonly used in practice within the field of farm management. However, productive efficiency measures were shown to be inappropriate to assess the overall economic performance of an enterprise (Musshof et al. 2009). Hence, we used here a classical profitability indicator from the field of farm management, namely the work income per full-time family work unit. It assesses the farm income available per unpaid full-time family labor force after equity capital has been remunerated to its opportunity cost. This latter is defined as the interests that would be generated would the equity capital be remunerated to the interest rate on ten-year Swiss government bonds. The work income per annual full-time family work unit was derived from the accountancy data of the farms.

2.5. Environmental performance

Relying on the considerations of Halberg et al. (2005), Jan et al. (2012a) distinguished between the local and the global environmental performance of a farm. The local environmental performance of a farm was measured by means of a so-called area-based[3] indicator assessing the amount of environmental impact generated on-farm (i.e., at the level of the local ecosystem of the farm) per hectare farm area. The global environmental performance of a farm was quantified by means of an eco-efficiency indicator, eco-efficiency[4] being defined as the farm agricultural output per unit of environmental impact generated in the "cradle-to-farm gate" link of the food chain.

For the same reasons as those exposed in Jan et al. (2012a), the present work concentrated on the global environmental performance of a farm. Eco-efficiency was specified here as the amount of digestible energy (in MJ) produced by the dairy farm per unit of environmental impact. Specifically, nine eco-efficiency indicators were estimated: one for each of the eight environmental impact categories considered (this first type of indicator being called partial eco-efficiency indicator as it considered only one environmental impact category at a time) and an aggregate eco-efficiency indicator considering several impact categories at the same time and estimated according to the Data Envelopment Analysis–based approach described in Jan et al. (2012a). The aggregate eco-efficiency was expressed in percent and can be interpreted as the result of a benchmarking in which a farm is compared to its peers in terms of digestible energy production per aggregate environmental impact. Further details on the interpretation of this aggregate eco-efficiency are available in Jan et al. (2012a). To better understand the effect of the factors considered on each partial eco-efficiency indicator, partial eco-efficiency was decomposed into its sub-constituents, namely partial eco-efficiency at the input-group (e.g., energy carriers, fertilizers) level defined as the amount of digestible energy produced by the farm per unit of environmental impact attributable to this input group. To facilitate the analysis, the partial eco-efficiency indicators were converted into environmental-intensity indicators, environmental intensity being defined as the inverse of eco-efficiency (Verfaillie and Bidwell 2000). This conversion enabled us to separate per input group the environmental impact generation per MJ digestible energy produced[5,6].

2.6. Analysis of the factors affecting environmental and economic performance

As mentioned in the introduction, the objective of the present contribution was to analyze the factors affecting the environmental and economic performance of Swiss dairy farms located in the alpine area. Numerous fac-

[3] Term used by Halberg et al. (2005).

[4] Eco-efficiency is the inverse of environmental intensity, also called product-based indicator by Halberg et al. (2005).

[5] The use of eco-efficiency for this decomposition would have been associated with problems of additivity between the average eco-efficiency at whole-farm level and its constituents at input-group level. These problems result from the fact that eco-efficiency, like any ratio variable, is undefined if its denominator, in the present case the environmental impact generation, equals zero. Such a situation is met often at input-group level. This problem does not occur when environmental intensity is used.

[6] Eleven input groups were defined: (i) fertilizers and nutrients, (ii) energy carriers, (iii) purchased animal feed, (iv) buildings and equipments, (v) machinery, (vi) plant protection products, (vii) purchased seeds, (viii) purchased animals, (ix) on-farm emissions from animals (stable), (x) own animals not present on the farm (e.g., outsourcing of heifers rearing, summering of cattle) and (xi) other inputs.

tors[7] can impact farm economic and environmental performance. These factors can be classified into two groups: factors pertaining to the general environment of the farm and those related to the farm itself as economic agent (Jan et al. 2011). The first group can be split up into three major sub-groups: the legal/regulatory environment, the socio-economic environment and the natural environment. The second group encompasses three sub-groups: the structural factors, the management factors and the human factors. Taking into account the variable availability and the limited sample size, we focused in the present work on the following factors: natural production conditions, farm size, farm type (full-time or part-time farm), production form and agricultural education of the farm manager. The natural production conditions were represented by the categorical variable "agricultural production zone," this variable being made of three modalities: hill zone, mountain zones 1&2 and mountain zones 3&4. The agricultural zone classification was based on criteria regarding (i) the climatic conditions and especially the vegetation period length, (ii) the accessibility in terms of transport and (iii) the topography (FOAG 2008). Within the mountain region, the favorableness of the natural production conditions decreases from mountain zone 1 to 4. Farm size was measured in terms of food production quantity defined as the amount of digestible energy produced by the farm. Farm type was represented by a dummy variable (full-time *versus* part-time farming). Full-time farms were defined as farms whose household income originated from at least 90% agricultural income. Part-time farms were farms with at least 10% of their household income originating from non-agricultural activities. Production form had two categories: organic *versus* conventional farming. The variable related to the agricultural education of the farm manager considered two levels: (i) completed apprenticeship or lower agricultural education level and (ii) agricultural education level higher than a completed apprenticeship (e.g., master craftsman diploma or university degree).

Taking into account the limited sample size as well as the number of independent variables analyzed and considering the requirements in terms of number of observations for performing a multiple linear regression analysis[8], we had to reject this multivariate approach, which would have suited best for the purpose of the present work. Instead, we investigated separately the effect of each factor on each performance indicator considered. Owing to the limited size of the sample and to the fact that the assumptions (*inter alia* normal distribution assumption) required for performing parametric tests were not fulfilled, this effect was investigated by means of non-parametric statistical tools. If the determinant was interval-scaled, we used the non-parametric Spearman's rank correlation to assess the relationship between this determinant and the performance indicator considered. In the case of a categorical determinant, its effect on the performance indicator was analyzed by means of a Mann-Whitney U-test if the factor in question had two categories or of a Kruskal-Wallis test if the factor considered had more than two categories.

Based on this analysis, we could identify the factors presenting synergies (i.e., influencing both the economic and environmental performance in the same direction) and those showing trade-offs (i.e., affecting the two dimensions in an opposite direction) in the enhancement of farm economic and global environmental performance.

3. Results

The results of the analysis of the factors affecting the economic and global environmental performance of Swiss dairy farms in the alpine area are presented in the following five sub-sections[9]. Each sub-section is devoted to one of the factors presented previously in section 2. Due to the high correlation of the environmental intensity between terrestrial and aquatic toxicity (Spearman's rho=0.91, $p<0.001$) and for the sake of conciseness of the paper, the results regarding the aquatic toxicity are not presented here.

[7] The terms "factor," "determinant," "independent variable" or "predictor" are used here as synonyms.

[8] Harrell (2001, p. 61) stated, as rule of thumb, that at least 10 to 20 observations should be available per predictor to obtain a reliable fitted-regression model. Applied to the present investigation, this rule would imply that at least 60 to 120 observations would be needed since the model encompassed six predictors and the categorical variable regarding the natural production conditions was transformed into two dummy variables.

[9] Owing to the paper length limit, we could not provide all detailed figures from the statistical analyses and tests carried out and had to focus on the most important results.

3.1. Effect of unfavorable natural production conditions

As obvious from Figure 1, the natural production conditions were shown to have a statistically significant effect on the aggregate eco-efficiency ($p<0.001$). Whereas farms located in the hill zone showed an average aggregate eco-efficiency of 86%, the average aggregate eco-efficiency of farms located in the mountain zones 1&2 and 3&4 was 68% and 36%, respectively[10]. With respect to the single environmental impact categories, we observed that the natural production conditions significantly influenced the environmental intensity regarding the demand for non-renewable energy resources ($p<0.001$), the global warming potential ($p<0.001$), the eutrophication potential ($p<0.001$), the terrestrial ecotoxicity ($p=0.06$), the human toxicity ($p<0.001$) and the land use ($p<0.001$). Farms located in the hill zone showed a significantly lower environmental intensity than those located in the mountain zones 1&2 and 3&4, the farms of the mountain zones 3&4 registering the highest environmental intensity for all impact categories considered. The environmental intensity differences between the three groups of zones were substantial and varied in their relative amplitude depending on the impact category considered. For example, the differences between the hill zone and the mountain zones 3&4 varied between a factor of 1.6 for the terrestrial ecotoxicity and 3.7 for the land use. A more detailed analysis at the input-group level showed that the differences were not due to a specific input group but concerned almost all input groups relevant for the impact category considered.

With respect to the economic performance, measured by the work income per family work unit, we observed significant differences between the three groups of zones considered ($p=0.07$, see Figure 2). The highest average income was observed for the hill zone (46'315 CHF per family work unit). The mountain zones 1&2 and 3&4 showed, compared to the hill zone, a lower work income per family work unit (29'930 and 34'550 CHF, respectively).

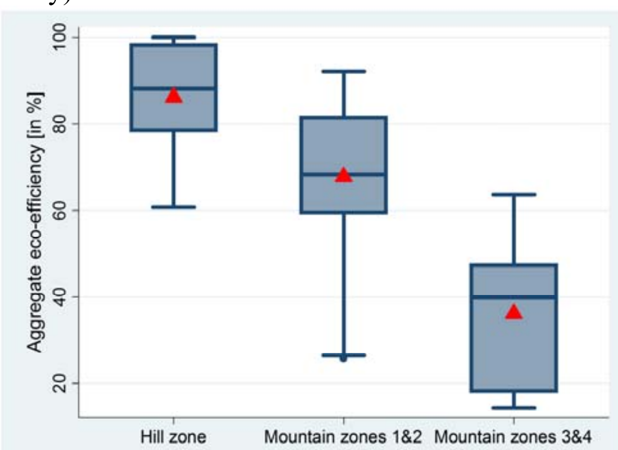

Figure 1. Effect of the natural production conditions on the aggregate eco-efficiency

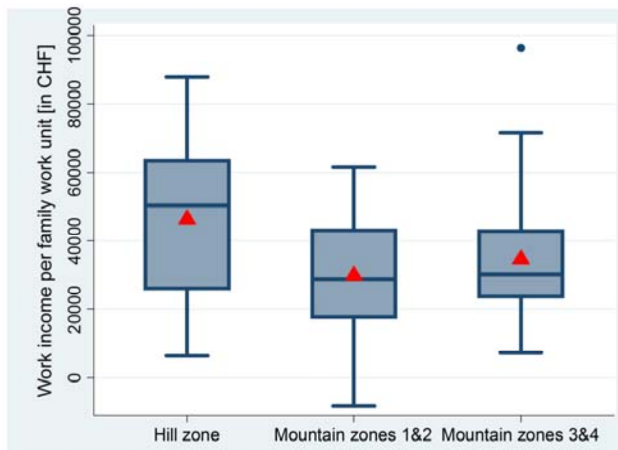

Figure 2. Effect of the natural production conditions on the work income per family work unit

3.2. Effect of farm size

The results of the non-parametric Spearman's rank correlation analysis showed the existence of a strong positive monotonic relationship between farm size, measured as farm digestible energy output, and farm aggregate eco-efficiency ($rho=+0.75$, $p<0.001$, see Figure 3).

The Spearman's rank correlation between farm size and environmental intensity was -0.73 ($p<0.001$) for the demand for non-renewable energy resources, -0.81 ($p<0.001$) for the global warming potential, -0.57 ($p<0.001$)

[10] All boxplot representations in this paper have to be interpreted as follows. The upper line of the box represents the upper quartile (Q75), the lower line of the box the lower quartile (Q25), the line subdividing the box the median (Q50). The lower whisker spans all data points within the range]Q25-1.5 IQR; Q25[(where IQR=interquartile range defined as Q75-Q25). The upper whisker spans all observations within the range]Q75; Q75+1.5 IQR[. Outliers (i.e., observations outside the whiskers) are marked as blue points. The red triangle represents the average.

for the eutrophication potential, -0.34 (p=0.01) for the terrestrial ecotoxicity, -0.70 (p<0.001) for the human toxicity and -0.80 (p<0.001) for the land use.

The negative correlation between farm size and environmental intensity proved to be related not to a particular input group but to the most important input groups for the environmental impact category considered.

A positive monotonic relationship was found between farm size and the work income per family work unit (rho=+0.37, p=0.005, see Figure 4).

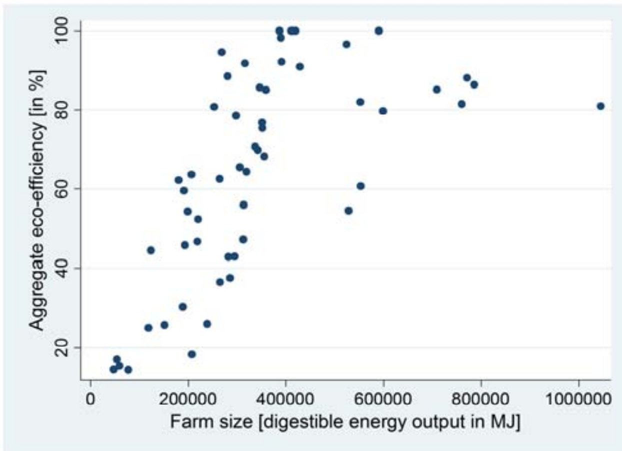

Figure 3. Effect of farm size on the aggregate eco-efficiency

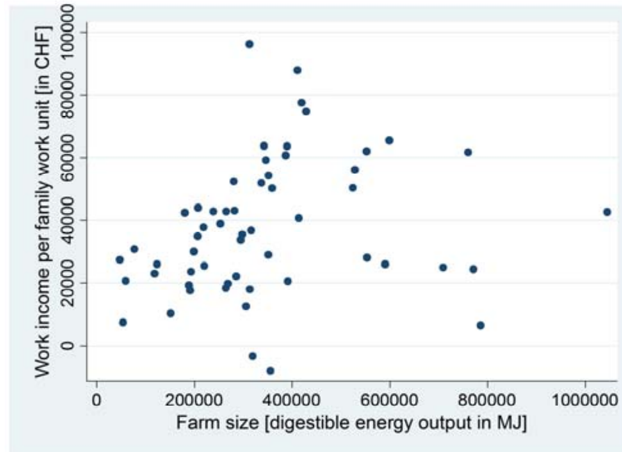

Figure 4. Effect of farm size on the work income per family work unit

3.3. Effect of farm type

Even if, based on a visual analysis of Figure 5, it would seem that full-time farms exhibited a higher aggregate eco-efficiency than part-time farms, the outcome of a Mann-Whitney U-test (p=0.15) showed that the differences were not significant at a 0.10 level. However, at the level of the environmental intensity for each single impact category, we observed some significant differences between these two groups of farms. Full-time farms tended to exhibit a lower environmental intensity than part-time farms regarding the demand for non-renewable energy resources (p=0.10), this being attributable to the input groups (i) purchased animal feed and (ii) buildings and equipments. With respect to the global warming potential, full-time farms also were characterized by a lower environmental intensity (p=0.04) being imputable to the input groups (i) purchased animal feed and (ii) buildings and equipments. In terms of eutrophication potential, full-time farms also showed a significantly lower environmental intensity than part-time farms (p=0.02). This better performance resulted from the input group fertilizers and nutrients. The lower environmental intensity of full-time farms compared to part-time farms observed for the impact category human toxicity (p=0.005) was ascribable to three input groups: (i) energy carriers, (ii) buildings and equipments and (iii) purchased animal feed.

In terms of economic performance, full-time farms significantly differed from part-time farms (p<0.001, see Figure 6). The average work income per family work unit was 51’614 CHF for full-time farms and 27’764 CHF for part-time farms.

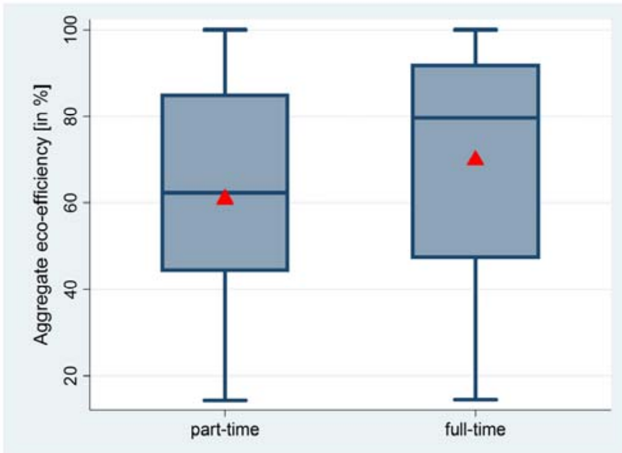

Figure 5. Effect of farm type on the aggregate eco-efficiency

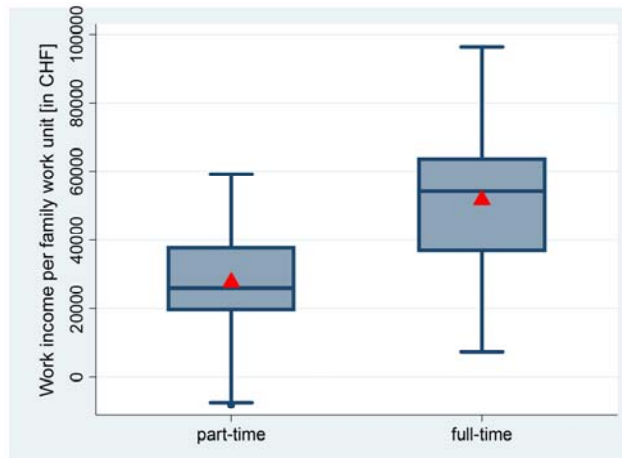

Figure 6. Effect of farm type on the work income per family work unit

3.4. Effect of production form

The boxplots of Figure 7 depict the variability of the aggregate eco-efficiency of the farms investigated for each production form (organic *versus* conventional farming). As apparent from this figure, organic farms showed a significantly higher aggregate eco-efficiency than conventional farms (p=0.06). The average aggregate eco-efficiency was 76% and 61% for the organic and conventional farms, respectively. A closer look at the environmental intensities revealed that—with the exception of the impact category land use—organic farms exhibited for all impact categories considered a significantly lower environmental intensity than conventional farms. Analyzing the environmental intensity at the input-group level, we found that the lower (p=0.09) environmental intensity regarding the demand for non-renewable energy resources was attributable to two input groups: (i) purchased animal feed and (ii) purchased animals. Two input groups accounted for the lower (p=0.06) environmental intensity regarding the global warming potential of organic farms: (i) purchased animal feed and (ii) fertilizers and nutrients. The better (p=0.02) environmental performance of organic farms in terms of eutrophication potential was found to result from their lower environmental intensity for the input groups (i) fertilizers and nutrients, (ii) purchased animal feed and (iii) purchased animals. With respect to the terrestrial ecotoxicity (p<0.001), the lower environmental intensity of organic farms could be ascribed to the input groups (i) purchased animal feed, (ii) plant protection products and (iii) purchased animals. Three input groups explained the better (p<0.001) performance of organic farms in terms of environmental intensity with respect to human toxicity: (i) energy carriers, (ii) purchased animal feed and (iii) purchased animals. Land use was the single impact category for which no significant differences in terms of environmental intensity was found between organic and conventional farming.

Compared to conventional farms, organic farms were characterized not only by a better environmental performance but also by a significantly better economic performance (p=0.01; see Figure 8). On average, the work income per family work unit was 53’712 CHF for organic farms compared to 32’676 CHF for conventional ones.

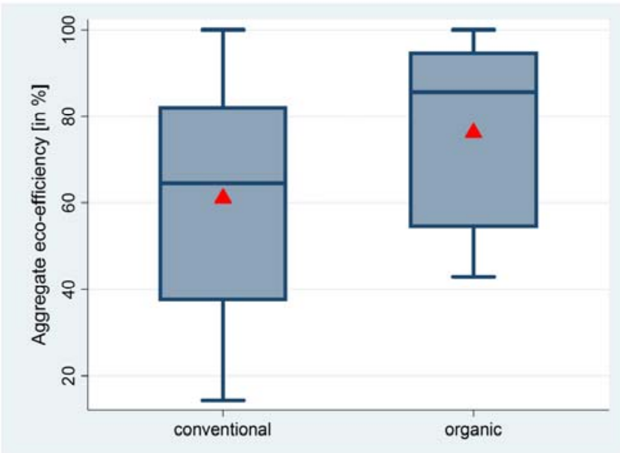

Figure 7. Effect of the production form on the aggregate eco-efficiency

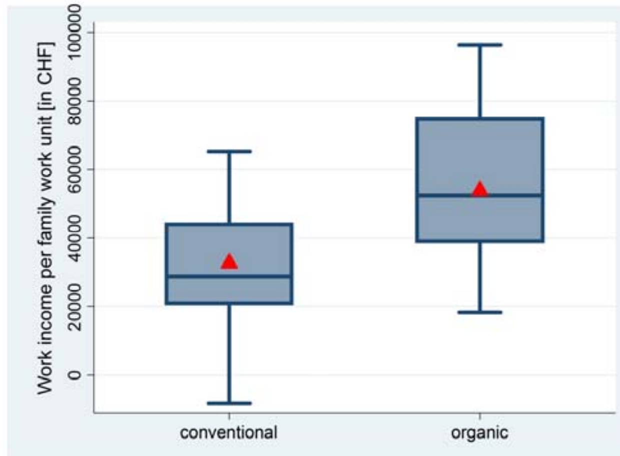

Figure 8. Effect of the production form on the work income per family work unit

3.5. Effect of agricultural education level

The effect of the agricultural education level on the aggregate eco-efficiency is depicted in Figure 9. A high agricultural education level was associated with a high aggregate eco-efficiency (p=0.009). The average aggregate eco-efficiency of farms whose managers had an agricultural education level higher than an apprenticeship was 77% compared to 57% for farms whose managers had an agricultural education level equivalent to or lower than an apprenticeship. A further analysis of the data revealed that a higher agricultural education involved a lower environmental intensity for all impact categories considered. This lower environmental intensity was not imputable to a particular input group but applied to almost all input groups relevant for the environmental impact category considered. A high agricultural education was revealed to be beneficial not only in terms of environmental performance but also in terms of economic performance (p=0.09, see Figure 10). The average work income per family work unit of a farm managed by a person with an agricultural education level higher than the apprenticeship was significantly higher than that of a farm managed by a person with an agricultural education level equivalent to or lower than the apprenticeship (44'439 CHF and 33'432 CHF, respectively).

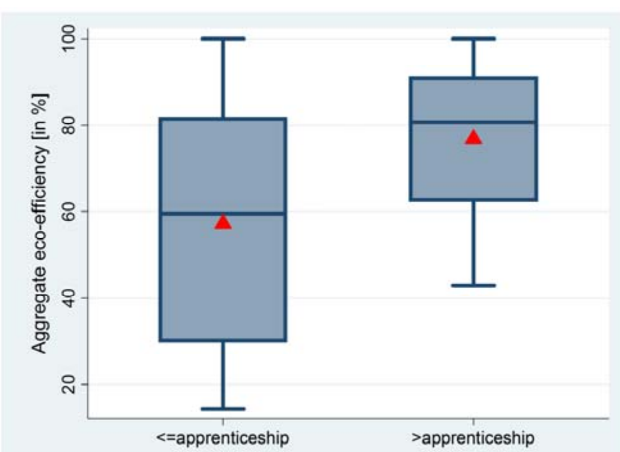

Figure 9. Effect of the agricultural education level on the aggregate eco-efficiency

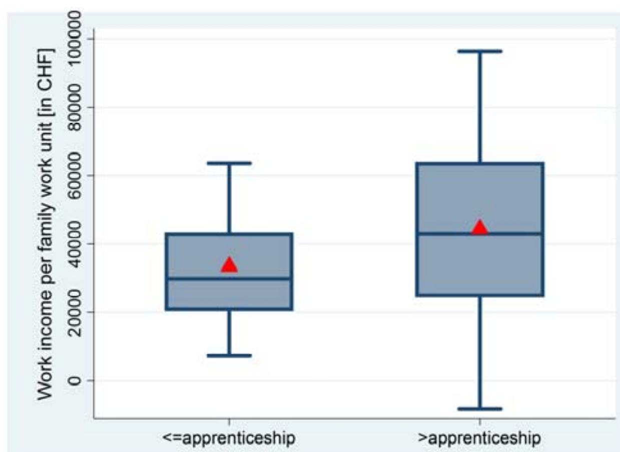

Figure 10. Effect of the agricultural education level on the work income per family work unit

4. Discussion

4.1. Main findings

The present work assessed the impact of selected factors on farm economic and environmental performance. Unfavorable natural production conditions were shown to strongly negatively influence both the global environmental and the economic performance of the farms investigated. The negative impact of unfavorable natural production conditions on farm eco-efficiency was not due to a specific input group but concerned all input groups. This finding highlights the systematic environmental but also economic competitive disadvantage of milk production under unfavorable natural production conditions. This finding also may question the appropriateness of milk production under such conditions. However, it should be considered here that dairy farms located in (unfavorable) mountain areas are characterized by their multifunctionality, that is, by their multiple functions going far beyond the simple food production function. The non-commodity outputs[11] that are associated with these additional functions and are by-products of the agricultural commodity outputs were not considered in the output variable used to assess eco-efficiency. This should be kept in mind when interpreting the results.

The result that farm size had a positive effect on the two considered dimensions of the sustainable performance of a farm emphasizes the substantial scale effects that exist in dairy farming regarding the use of not only economic but also environmental resources.

Compared to full-time farming, part-time farming, which was shown for Switzerland (see Jan et al. 2011; Roesch 2012) to be associated with a substantially lower economic performance, herein was also synonymous with a poorer global environmental performance. This seemed to result primarily from an inefficient use of purchased feed, of buildings and equipments and, in some cases, of fertilizers and energy carriers.

Whereas organic farming was shown in the literature to be associated with a—compared to conventional farming—higher eco-efficiency for some impact categories and with a lower one for others (see Tuomisto et al. 2012), in the present work it was associated with a higher eco-efficiency for all impact categories with the exception of land use, for which no significant differences were observed between organic and conventional farms. This higher eco-efficiency was attributable to the input groups purchased animal feed, fertilizers and nutrients and purchased animals[12], highlighting thus that this better performance resulted primarily from the feeding and fertilization strategies and practices of organic farming. This finding implies that—under the natural production conditions of the alpine area and the associated production restrictions and low forage yield potential—organic farming may be from both an environmental and an economic perspective a more appropriate technology than conventional farming for the dairy activity. Thus, a process of conversion from conventional to organic farming may be very likely to lead to economic and environmental benefits and to a substantial improvement of the sustainability of the dairy food chain in this region. This probably explains why the share of organic farms increases with the unfavorableness of the natural production conditions (e.g., in 2012, according to the Swiss Federal Statistical Office, the proportion of organic farms in the mountain zone 4 and in the plain zone was 35% and 5%, respectively).

Last but not least, agricultural education was shown to play an important role in terms of environmental and economic performance. A high agricultural education implied both high eco-efficiency and high work income per family work unit. All input groups were involved in this high eco-efficiency suggesting that better educated farm managers had better management capacities for the use of economic and environmental resources.

4.2. Limits of the study

For the interpretation and discussion of the results of the present investigation as well as their implications, attention should be paid to the following issues. Firstly, it should be noted that the investigation focused on only one of the two dimensions of the environmental performance of a farm, namely the so-called global environmental performance defined by Jan et al. (2012a) as the eco-efficiency of food production until the farm gate. The

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[11] These non-commodity outputs, such as conservation of a mosaic rural landscape or contribution to the vitality of rural communities, are by-products of the agricultural commodity outputs.

[12] Some readers may be surprised that plant protection products are not mentioned here. This is because the present contribution focuses on farms located in the alpine area, a region in which the use of plant protection products in conventional farming is very limited and very often inexistent.

local environmental performance of a farm, which focuses on the environmental impact generation that occurs locally at the farm level and which is defined as the environmental pressure exerted by the farm on its local ecosystem, was not investigated in the present work. It is therefore necessary to remember that the findings of the present contribution regarding the factors affecting farm environmental performance apply to only the global dimension of farm environmental performance. Secondly, the sample was not selected at random due to the comprehensiveness and complexity of the data collection. This may have introduced a positive bias in the representativeness of the sample as it has to be expected that farm managers interested in environmental issues were more likely to participate in the project than those who did not feel concerned by such issues. Thirdly, an additional sample-related limitation of the investigation lies in the approach used to assess the effect of the selected factors on farm environmental and economic performance. As mentioned in section 2.6, due to the limited sample size, we had to refrain from applying multiple linear regression analysis and therefore investigated separately the effect of each factor on each performance indicator. Consequently, the effect measured for each factor is not a *ceteris paribus* one and may capture the effects of other factors correlated with the one investigated.

5. Conclusion

The present contribution provides initial evidence that the promotion of an economically viable alpine dairy farming sector as well as the enhancement of one with a high eco-efficiency are not antinomic but synergetic. By increasing farm size (i.e., through scale effects), by promoting organic farming as well as full-time farming and by raising the level of agricultural education among future farm managers, major enhancements in terms of economic and environmental performance could probably be achieved. Our work shows how valuable combined micro-level economic and environmental data are. Such data enable us to gain a better insight into the relationship between these two dimensions of sustainability. Such insight is a pre-requisite if we wish to improve sustainability. In order to gain a holistic understanding of the farm-level link between economic and environmental performance, our future studies will focus on the analysis of the link between (i) economic performance and local environmental performance and (ii) local and global environmental performance.

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Analysis of inconsistencies between Product Category Rules in the same supply chain – a case study of food PCRs

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ABSTRACT

According to ISO 14025, Product Category Rules (PCRs) are developed as the basis for environmental declarations. The PCRs define the requirements for the underlying life cycle assessment and the format for reporting in an Environmental Product Declaration (EPD) for a given product category. Because PCRs are developed separately by the interested stakeholders, one risk of this approach is that similar methodological aspects could be treated differently in different but related PCRs. The International EPD System has developed a network approach for food PCRs to avoid inconsistencies by 1) creating one document identifying issues common to all food PCRs; and 2) simplifying the development of new PCRs by permitting inclusion by reference of aspects treated in other PCRs. It may be appropriate to extend this network methodology also to other product groups and sectors, and to extend the co-ordination across program operators.

Keywords: Product Category Rules, Food, Environmental Product Declaration, PCRs management, ISO 14025

1. Introduction

Product Category Rules (PCR) define the requirements for the underlying life cycle assessment and the format for reporting in an environmental declaration type III for a given product category. PCR documents are developed within a program for environmental declarations type III according to ISO 14025 (ISO, 2006).

Although the details regarding the process might differ somewhat between programs, PCRs in the same program are normally drafted by the stakeholders interested in the product group. Because PCRs may have scopes covering intermediate or final products, different PCRs may include the same life cycle stages and associated methodological guidance. There is thus a risk that similar methodological aspects are treated in different ways in related PCRs. An example could be wheat that is used an ingredient for many food products such as pasta and bakery products. If the calculation rules for the wheat cultivation process are described separately in the PCR for pasta and the PCR for bakery products, some key methodological aspects shared by the two product systems may be managed in different ways.

The International EPD System is a program for type III environmental declarations with a broad scope in terms of product categories and geographical area covered. As of April 2014, there are 108 PCRs registered within the program, whereof 27 belong to the category “Food and agricultural products” (International EPD System, 2014). The PCRs within this category have been developed in a time period of several years and by different stakeholders. All PCRs fulfill the program instructions of the International EPD System but within the PCRs, in some cases, different approaches have been taken, leading up to inconsistencies. To avoid anticipated inconsistencies between related PCRs, a pilot project called the “Network of PCRs for Food” was initiated in 2013 with the scope of agricultural and food products (International EPD System, 2013b).

2. Methods

This paper identifies and analyses the methodological inconsistencies between PCRs within the International EPD System and how they have been treated in the first year of the network of PCRs for food pilot project. The method chosen is a qualitative identification of the key methodological questions and a comparison of the different guidance given in the PCRs before and after the alignment. The main LCA methodological aspects that are normally covered by PCRs include (International EPD System, 2013):

- Functional unit
- System boundary
- Allocation procedure
- Data quality requirements

– Cut-off rules
– Data collection and calculation procedures
– Impact assessment categories
– Scenario assumptions

The analysis focuses on these methodological aspects and on seven linked PCRs available within the food and agricultural products category of the International EPD System; see Table 1. PCRs and related guidance documents from other programs and initiatives were not included in the scope of the analysis.

Table 1. Food and agricultural PCRs in the International EPD System included in the analysis.

Name [a]	PCR registration number	Version
Arable crops	2013:05	1.01
Raw milk	2013:16	1.01
Processed liquid milk and cream	2013:17	1.01
Yoghurt, butter and cheese	2013:18	1.01
Meat of mammals	2012:11	2.0
Grain mill products	2013:04	1.02
Uncooked pasta	2010:01	2.0

The identified connections between the different PCRs are illustrated in Figure 1. There are three levels of the PCRs, with the ones earlier in the supply chain function as the upstream system for the following two levels. The PCR for arable crops (including wheat) is part of the shared upstream supply chain for all the other analyzed PCRs, including milk, meat and grain mill products. The PCR for raw milk relates to the animal husbandry, which is the same technical system as is described in the PCR meat of mammals. From an LCA perspective, beef and raw milk are two co-products from the same process, which results in an allocation problem to be handled in the PCR. The different dairy products are covered by two different PCRs, which share a similar allocation problem between different co-products from the dairy plant.

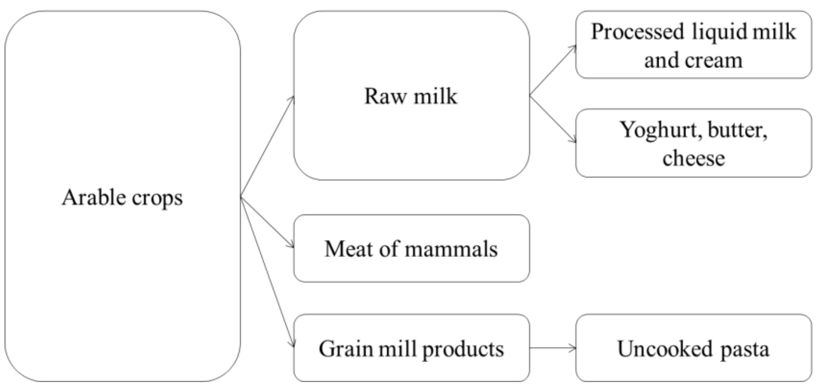

Figure 1. Identified supply-chain connections between the analyzed PCRs.

3. Results

3.1. Identified inconsistencies

The qualitative assessment identified that potential methodological aspects were those normally specified in a PCR: system boundaries, data quality requirements, allocation procedures and allocation factors. Aspects such as the choice of systems approach (e.g. attributional or consequential LCA) and environmental impact categories were less relevant in the analyzed PCRs because the basic requirements for these aspects are set up in the General Program Instructions and thus shared between all PCRs (International EPD System, 2013a). The main methodological aspects with potential inconsistencies that could alter the results among the analyzed PCRs were identified as:

– factors used for the estimation of emissions from fertilizer use (EF),
– allocation between agricultural co-products (AG),
– allocation between milk and meat (MM),

– allocation among co-products at the mill (MI),
– allocation among co-products at dairy plant (DP).

Table 2 lists all the PCRs in which the analyzed factors were treated. Some of the aspects (EF and AG), were treated in all the analyzed PCRs, while other were only treated in some of the analyzed PCRs.

Table 2. Main methodological aspects and PCRs where they were treated – before the network of PCRs.

	Arable crops	**Raw milk**	**Processed liquid milk and cream**	**Yoghurt, butter, cheese**	**Meat of mammals**	**Grain mill products**	**Uncooked pasta**
EF	X	X	X	X	X	X	X
AG	X	X	X	X	X	X	X
MM	-	X	X	X	X	-	-
MI	-	-	-	-	-	X	X
DP	-	-	X	X	-	-	-

These methodological aspects may greatly affect the environmental impact of a product. As an example the results for climate change (calculated with Global Warming Potential, 100 years) of wheat grain and straw as a function of different methods of allocation was calculated (Figure 2). While the results may vary greatly between different cases and methodological aspects, the example shows that the results have the potential to vary significantly (here: -45% for mass allocation compared to the case where straw is considered as waste).

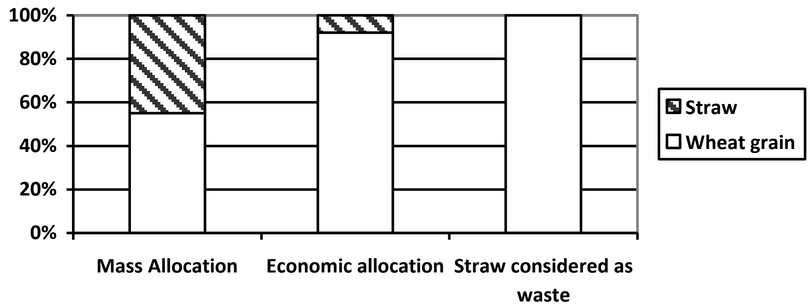

Figure 2. Normalized example results for the climate change indicator results (calculated with GWP, 100 years) of wheat grain and straw as a function of different methods of allocation. The result of straw considered as waste has been set to 100%.

An EPD in the International EPD System uses as a default life cycle inventory (LCI) indicators for resources use and four environmental impact categories (International EPD System, 2013a). Table 3 shows which methodological aspects affect which indicator results and environmental impact categories.

Table 3. Main methodological aspects and potentially affected indicator results and environmental impact categories.

Factor	**Resource use (LCI)**	**Climate change**	**Acidification**	**Eutrophication**	**Photochemical oxidant formation**
EF	-	X	X	X	-
AG	X	X	X	X	X
MM	X	X	X	X	X
MI	X	X	X	X	X
DP	X	X	X	X	X

Four of the five analyzed factors (AG, MM, MI and DP) have an influence of all the indicator results declared in an EPD, since different methods of allocation lead to different life cycle inventories and resulting environmental burdens attributed to the product object of the EPD. The other factor, EF, influences mainly the impact categories Climate change, Eutrophication and Acidification.

To avoid inconsistencies within the network it was decided to treat each of the methodological aspects in a single PCR and make reference to that in the other PCR documents (see Table 4). An example of this approach is that in the PCRs for arable crops rules for EF and AG are given. In the other PCRs in the upstream phase, the guidance on these methodological aspects is replaced by references to the PCR for arable crops.

Table 4. Main methodological aspects and in which PCRs they were treated – after the network of PCRs.

Factor	Arable crops	Raw milk	Processed liquid milk and cream	Yoghurt, butter, cheese	Meat of mammals	Grain mill products	Uncooked pasta
EF	X	-	-	-	-	-	-
AG	X	-	-	-	-	-	-
MM	-	X	-	-	X	-	-
MI	-	-	-	-	-	X	-
DP	-	-	X	X	-	-	-

In some cases, such as those of MM and DP, it was not possible to treat an aspect in only one PCR document, because some the PCRs involved (in this case Processed liquid milk and cream and Yoghurt, butter and cheese) are “parallel”, that means that they are at the same level of the supply chain and describe two different co-products from the same process. In such cases the coordination of the network ensured that these aspects are treated in the same way in both PCRs. The motivation to keep two different PCRs is that there is still a need for separate guidance for aspects such as the functional unit of the two different products.

3.2. Alignment of inconsistencies

Another issue was how to treat the aspects, since different approaches were possible and were already used in some PCRs. In Table 5 the analyzed and the chosen approaches are reported.

Table 5. Approaches used in the PCRs before the network, and the chosen approaches after the implementation of the network.

Factor	Approaches used before the network of PCRs	Chosen approach
EF	1. Data from literature (global) 2. Data from literature (regional) 3. Primary data	Data from literature (global) in absence of primary data.
AG	1. Economic allocation 2. Physical allocation (mass) 3. By-products considered as waste	Economic allocation
MM	1. Economic allocation, 2. Physical allocation (mass)	Economic allocation
MI	1. Economic allocation, 2. Physical allocation (mass) 3. By-products considered as waste	By-products considered as waste
DP	1. Economic allocation 2. Physical allocation (wet mass, dry mass, mass of protein and fat) 3. By-products considered as waste	Physical allocation (mass of protein and fat)

The approaches to be used were chosen after an open consultation with the interested stakeholders when each PCR was updated. The choice of the approaches was made according to the main literature and to the experience of the involved experts.

4. Discussion

The International EPD System’s food PCRs network is the first known example of increased coordination of PCRs in an ISO 14025 program. The results of the present study demonstrate that without coordination there would be many inconsistencies, which would make an objective evaluation of different products environmental performances impossible.

The problem of LCI modelling inconsistencies is not only relevant within the same program. Different ISO 14025 programs have chosen different methodological bases and product group classification systems that may or may not be compatible. There are also many initiatives to develop PCRs in programs similar to, but not based on, ISO 14025, such as the European Commission Product Environmental Footprint.

To put a similar coordination in place among different ISO 14025 programs is a big challenge but it certainly should be a target for which to aim.

5. Conclusion

The International EPD System's Network of PCRs has two main positive effects: avoiding methodological inconsistencies between PCR documents (by treating issues common to different PCRs in only one PCR document) and simplifying the development of new PCRs (by allowing incorporation by reference to the relevant PCR document). These benefits also translate to positive effects in the development of environmental declarations: the comparability is strengthened and the time needed to develop the first declaration in a product category is reduced.

For these reasons it may be appropriate to extend the methodology of the network to other product groups and sectors, for example construction products, wood and paper products or textile products, and to extend the co-ordination across ISO 14025 program operators and incorporated into initiatives independent from ISO 14025 such as the European Commission Product Environmental Footprint (EC, 2013). The insights gained on the need for alignment of PCRs in the same supply chain could also be relevant for guidance documents and international standards related to PCRs, such as the Guidance for Product Category Rule Development (Ingwersen and Subramanian, 2013).

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ONE TWO WE – Life cycle management in canteens together with suppliers, customers and guests

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ABSTRACT

The environmental impacts of all food purchases of the Swiss canteen operator SV Group were analysed within an LCA study. Improvement potentials were identified, which include measures in the canteen operation (e.g. reduction of food waste), measures in the supply chain (e.g. a reduction of vegetables grown in heated greenhouses) and dietary measures such as a reduction of the average amount of meat per meal. The results have been used to initiate the programme "ONE TWO WE" together with WWF Switzerland. It assists the customers (companies who commission the SV Group with the operation of canteens in their premises) to reach improved levels of environmental performance e.g. a 20% cut on GHG emissions in the supply chain.

Keywords: canteen, diet, gastronomy, life cycle management, supply chain, organisational environmental footprint

1. Introduction

Nutrition causes about 12% of total energy demand and 18% of greenhouse gas emissions due to Swiss consumption patterns (Figure 1). There are many other environmental impacts that should be accounted for in an LCA. Here we use the Swiss ecological scarcity method 2006 which weights different environmental indicators according to the political targets in Switzerland (Frischknecht et al. 2009). Environmental impacts are quantified as eco-points (UBP for Umweltbelastungspunkte). If all types of environmental impacts are included in the analysis with this method this share rises to about 30% (Jungbluth et al. 2011; Jungbluth et al. 2013). This is due to specific environmental impacts caused by agricultural practice such as pesticide use, heavy metal emissions from fertilizers, land and water use as well as problems caused by acidification and nutrification.

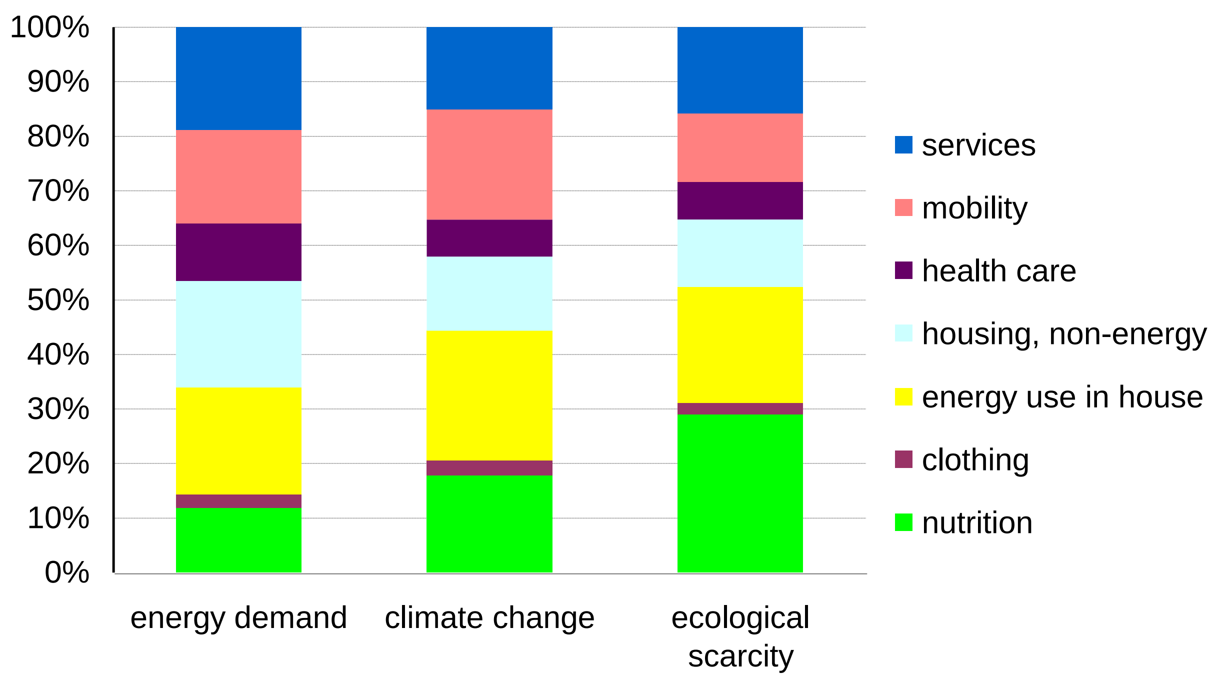

Figure 1. Importance of nutrition in total consumption (Jungbluth et al. 2011; Jungbluth et al. 2013)

The main part of the environmental impact arises from the agricultural production of meat and fish (Figure 2).

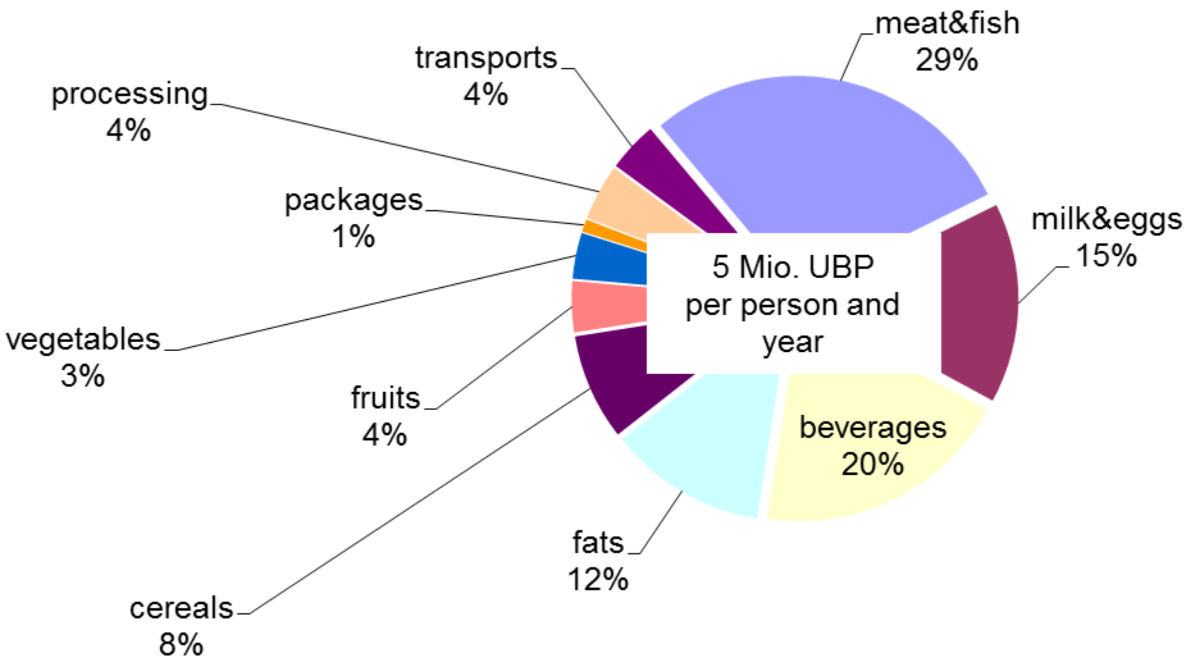

Figure 2. Importance of product groups in total impacts of nutrition. The total impact is 5 million eco-points (UBP – Umweltbelastungspunkte) per year

Environmental awareness is increasing in the gastronomy sector. The Swiss canteen operator SV Group commissioned ESU-services Ltd. to make an LCA of its full food and non-food purchases in order to identify improvement potentials. The LCA results are the basis of the program "ONE TWO WE", which aims to assist the customers of SV Group with the reduction of their GHG emissions by 20 %. The customers are companies that commission the SV Group with the operation of their canteen in their premises. The program is elaborated in close collaboration with WWF Switzerland and ewz Zürich (public electricity supply in the Swiss city of Zurich).

2. Methods

The environmental impacts of all food purchases in 240 canteens of the SV Group were analysed within an LCA study for the operation in the year 2011. In this year 19.2 million meals were prepared and sold. About 820 grams of food items, 2.2 dl (deciliter) of beverages and 40 grams of other materials such as serviettes were used per meal (Table 1).

Table 1. Composition of the average meal in canteens of the SV Group

	Weight in average meal (grams per meal)
meat/poultry	108
fresh vegetables	21
bread	310
dairy products	108
eggs	135
fish	5
sweets	137
convenience	32
beverages	225
non-food	42

The SV group provided a detailed list of their food purchases including weight and costs of the purchased items. In the next step, LCI data for 160 different food items available within the ESU food database (Jungbluth et al. 2014) were linked to the purchased amounts. Therefore also rough assumptions on packaging and transportation have been made for the different food items. This followed the idea of a modular LCA as developed in a previous research work (Jungbluth 2000).

The objectives of this project were twofold. In a first step, the most important ingredients were identified and the impacts of the food supply were compared with the direct global warming impacts of the canteen operation. Impacts of canteen operation (e.g. electricity and water use) were evaluated in previous years within the environmental reporting of the company (SV (Schweiz) AG 2010, 2008). The total impacts have been divided by the number of meals sold as a functional unit. Food waste was already monitored as part of this environmental reporting. Impacts due to its disposal are included in the assessment. The production of the wasted food is included in the figures on total food supply.

In the second stage of the project, improvement potentials were identified in the supply chain and the operation of the canteen. The results in this study are analysed across a representative range of impact categories with the ecological scarcity method (Frischknecht et al. 2009) and with Global Warming Potential (GWP) (Solomon et al. 2007).

3. Results

3.1. Global warming potential of average meal

The food purchases of the canteen operator were summarized for the categories of meat, fish, dairy products, eggs, vegetables, fruits, bread, sweets, beverages and convenience products. Within the group of convenience products a range of different types of food can be found. The contribution of each life cycle stage to the GWP of all food purchases is shown in Figure 3. The GWP is expressed per meal, which means that the GWP of all food purchases were divided by the total amount of meals delivered per year. The life cycle includes the production, the processing, the packaging, the transport of food items to the canteen and the operation (meal preparation at the canteen).

An average meal served in a canteen operated by the SV group has an average GWP of 4.1 kg CO2-eq. The agricultural production step is responsible for 60% of the emissions, the processing 8%, the packaging 2%, the transport 5% and the operation of the canteens (cooling, cooking, etc.) 25%. The overall GWP of the food supply is dominated by the meat and poultry products (35%), the dairy products (15%), the fresh vegetables (14%) and convenience products (14%).

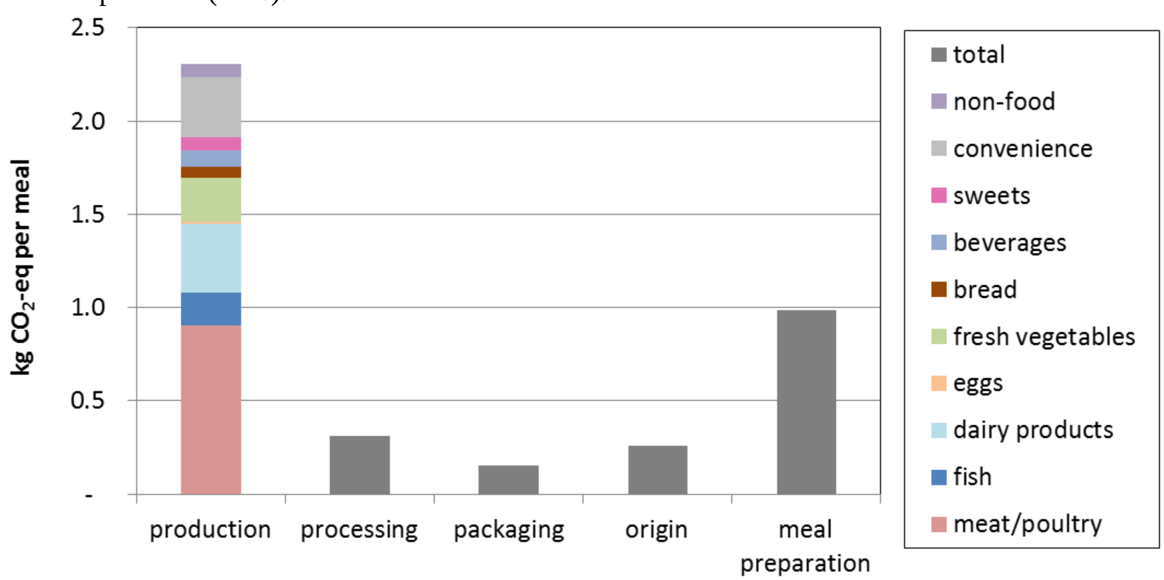

Figure 3. Global warming potential per meal of food purchases and canteen operation

3.2. Environmental impacts of average meal

The importance of the production of food products gets even more pronounced when total environmental impacts are evaluated according to the ecological scarcity method 2006 (Frischknecht et al. 2009). With this about 71% of total impacts are due to production of food products. The impacts per canteen meal are about 9'300 eco-points.

The environmental impacts of this meal can be compared with the average impacts due to nutrition in Switzerland. About 15'300 eco-points are caused due to the purchase of food products, but not including the consumption of food in canteens and restaurants nor including the delivery of food products to the home (Jungbluth et al. 2012). Thus, the canteen meal has quite some relevance which can also be partly explained by the higher share of meat (37%) than in the average daily consumption basket shown in Figure 4.

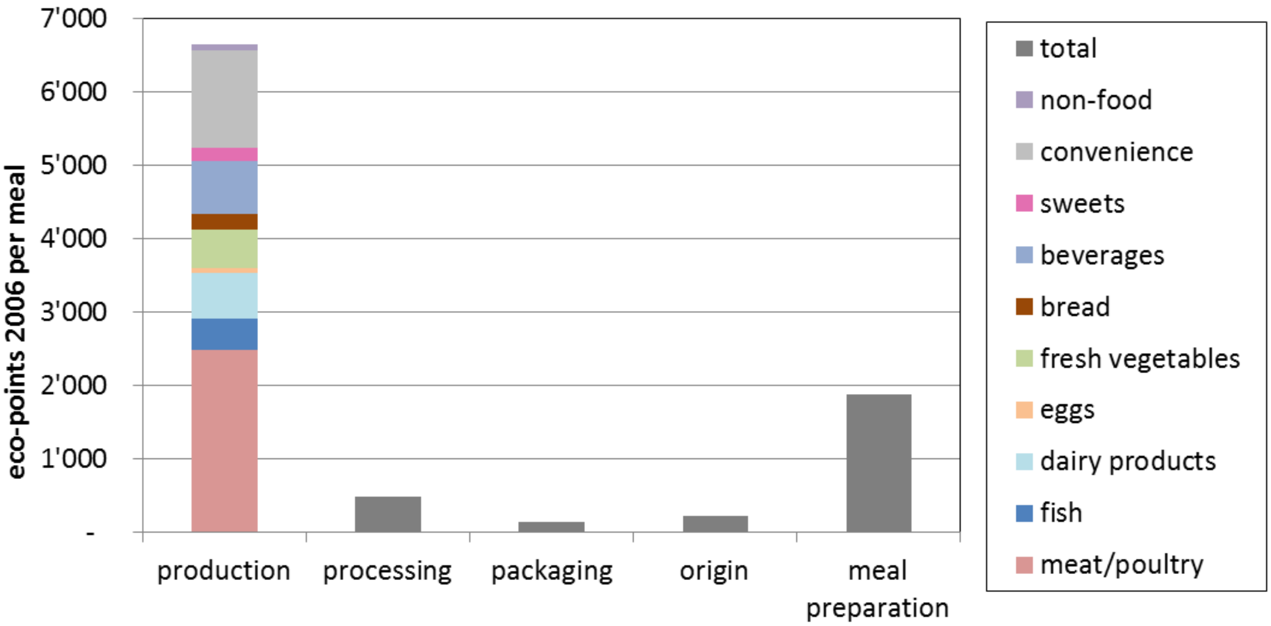

Figure 4. Eco-points according to ecological scarcity 2006 of food purchases per meal

The share of different types of environmental impacts for the different food categories is evaluated in Figure 5. The use of plant protection products, eutrophication, human health impacts due to ammonia emissions are important environmental aspects while evaluating the total food purchases of the canteen operator.

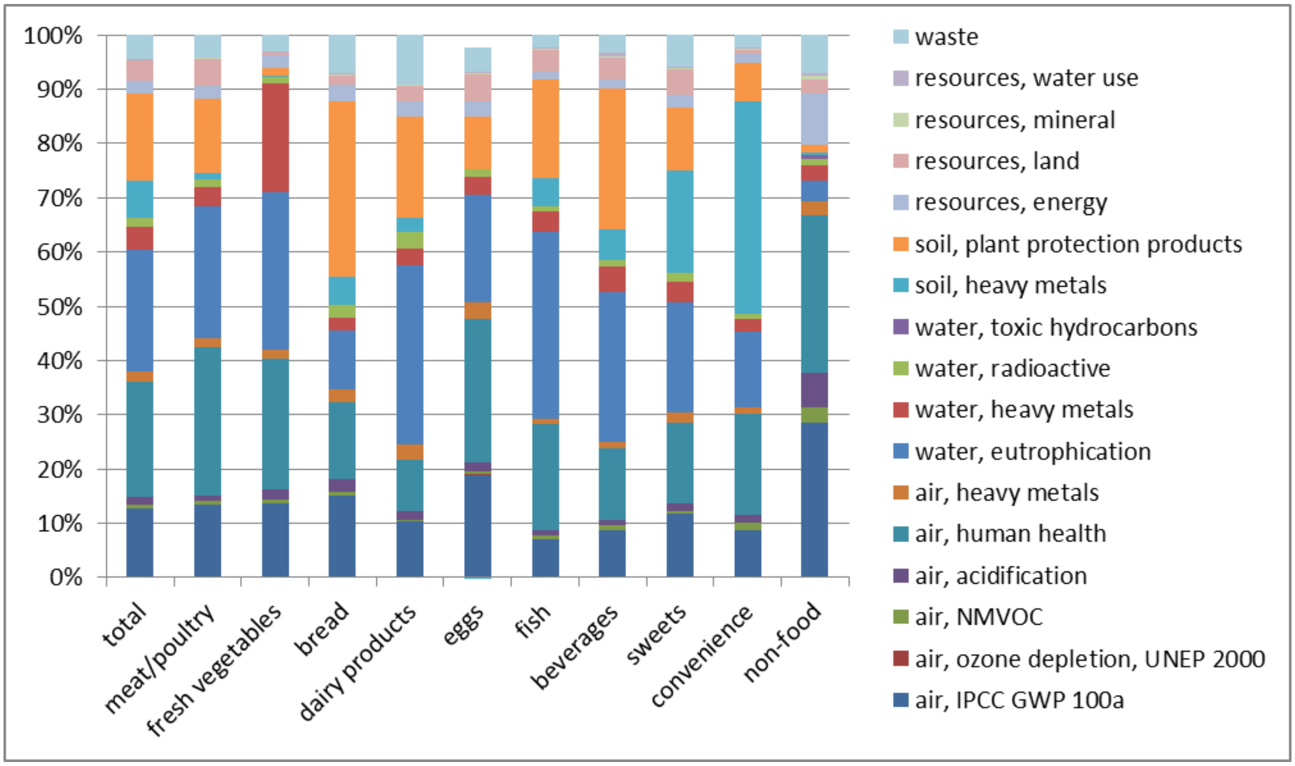

Figure 5. Share of category indicators according to ecological scarcity 2006 per meal of food purchases

3.3. Seasonal calendar to assist supply chain management

Based on the detailed assessment of the total environmental impacts, several improvement options for the supply chain have been discussed with the canteen operator. The detailed analysis of vegetable and fruit purchases has shown that an important part of impacts is due to the heating of greenhouse with fossil fuels when products are bought outside the Swiss season. Also transports by airplane can than play an important role. Therefore one option is the optimized purchase of fruits and vegetables.

The environmental impacts of vegetable and fruit purchases depend on the production period, the origin and the means of transport. For a given fruit or vegetable, all monthly supply routes were assessed in a seasonal calendar in order to provide better guidance for purchases. Two examples are illustrated in Figure 6.

For example, fresh broccoli is supplied from Switzerland, Spain and Italy. The fresh broccoli from Switzerland is only supplied from May to October and its GWP is 0.6 kg CO2-eq per kg. From January to May and from November to December, fresh broccoli is transported in truck from Spain and Italy. According to the supplier of the canteens, the fresh broccoli is produced in fossil heated greenhouses and its GWP is 7.2 kg CO2-eq per kg in January, February and December. The production of fresh broccoli that is deep-frozen in order to maintain a supply during the off-season generates a GWP of about 0.7 kg CO2-eq per kg. This increases with the time of storage after harvesting and is thus highest in April. Deep-frozen vegetables are an interesting alternative to fresh vegetables cultivated in heated greenhouses. The information concerning heating does not necessarily match with information found in literature about this type of vegetables (e.g. Milà i Canals et al. 2008).

Another relevant example is the supply of green asparagus. From July to February, green asparagus cultivated in Peru and transported by air cause a GWP of 12.8 kg CO2-eq per kg. Green asparagus cultivated in Switzerland or Spain and supplied from April to June have an average GWP of 1.6 kg CO2-eq per kg.

The cooperation with the wholesale dealer for vegetables shows that so far they often do not know in detail about the origin and way of production. A system to better monitor the important factors such as type and amount of heating for greenhouses for a given vegetable in a given months shall be build up in future.

kg CO2-eq per kg good		Jan	Feb	March	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Broccoli	CH-Lorry	n.a.	n.a.	n.a.	n.a.	0.6	0.6	0.6	0.6	0.6	0.6	0.6	n.a.
	ES-Lorry	7.2	7.2	0.9	0.9	0.9	n.a.	n.a.	n.a.	n.a.	n.a.	0.9	7.2
	IT-Lorry	7.1	7.1	0.7	0.7	0.7	n.a.	n.a.	n.a.	n.a.	n.a.	0.7	7.1
Broccoli deep frozen	CH-Lorry	0.70	0.72	0.74	0.77	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.68
Green asparagus	CH-Lorry	n.a.	n.a.	n.a.	1.5	1.5	1.5	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
	ES-Lorry	n.a.	n.a.	n.a.	1.7	1.7	1.7	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
	PE-Air	12.8	12.8	n.a.	n.a.	n.a.	n.a.	12.8	12.8	12.8	12.8	12.8	12.8
	US-Air	n.a.	9.7	9.7	9.7	9.7	9.7	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

Figure 6. Example of broccoli and green asparagus for the creation of the Season table (country codes, CH-Switzerland, ES-Spain, IT-Italy, PE-Peru, US-United States

4. Discussion and follow up

The programme "ONE TWO WE" was elaborated based on the results of the LCA and further collaborations. It consists of a set of improvement options in five fields namely the logistic, the canteen operation, the food supply and the food range (see Figure 7). Therefore targets on certain key performance indicators have been set.

The environmental performance of the logistic shall be improved by reducing the share of air-freight. The optimisation at the canteen includes for example the amount of food waste and energy efficiency (cooling, lighting, cooking and ventilation). The mitigation of the environmental impacts of the food supply relies on the reduction of fruit and vegetables cultivated in heated greenhouses based on a seasonable table which calculated the carbon footprint per month of different products. Another important measure is the reduction of the average quantity of meat per meal by offering attractive vegetarian meals and meals with a lower amount of meat per serving. Therefore an education program has been initiated in order to teach the cooks attractive ideas for vegetarian meals that can be prepared for canteens.

A good communication with the guest and customers should explain the background of this programme while at the same time allowing the guest to choose from attractive recipes. The program "ONE TWO WE" aims for a reduction of 20% on greenhouse gas emissions in canteens which follow all suggestions for improvements. The achieved reductions are documented transparently. Therefore a simplified Excel tool has been elaborated which allows a simplified calculation of greenhouse gas emissions per meal passed on some key environmental performance indicators to be entered by the canteen operator.

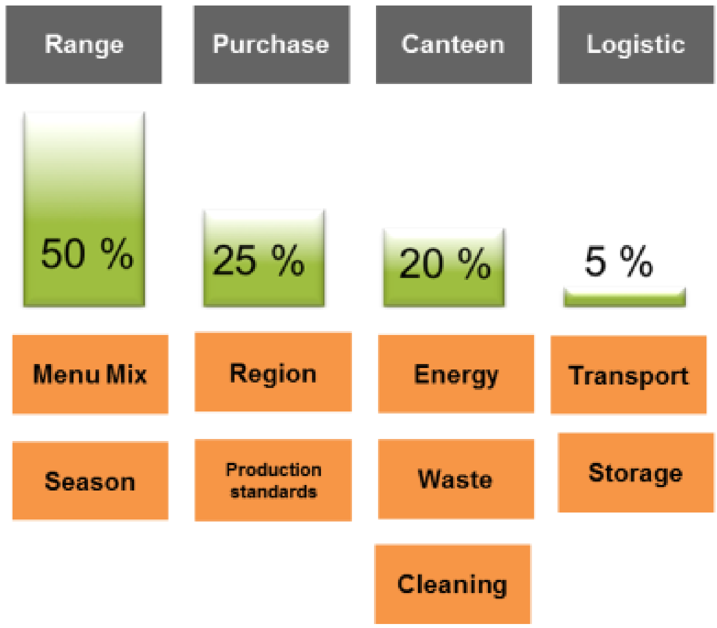

Figure 7: The program “ONE TWO WE” is structured in different steps

5. Conclusions

The programme “ONE TWO WE” started successfully with many customers positively convinced by the proposed changes in the provision of canteen meals. The program was launched in October 2012 in Zürich and is being implemented in 70 restaurants until the end of 2013. In 2013 the initiative " ONE TWO WE" has been awarded with the Zurich Climate Prize 2013. The future will show whether also the guest in the canteens support the changes.

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Evaluation of beef sustainability in conventional, organic, and mixed crop-beef supply chains

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ABSTRACT

Demand for sustainable meat with balance between environmental, economic and social (EES) aspects has increased recently. Brazil has promoted production of a wider variety of beef products and European Union is one of the target markets for Brazilian organic beef. This study aims to evaluate the EES performance of conventional and organic beef production. Accordingly, global warming, land occupation, energy consumption, operating profit and animal welfare were evaluated for different systems. Relevant indicators were measured using different methods, such as life cycle assessment, operating profit analysis, price volatility analysis, and qualitative scoring. Results confirmed a higher impact of organic beef production on global warming and land use due to animals' longer grazing period. Animal welfare, however, scored slightly higher for the organic beef production systems. Organic beef showed a slightly higher profitability, but its economic sustainability is constrained by technical barriers and higher transaction costs.

Keywords: Organic, conventional, mixed crop-beef, life cycle assessment

1. Introduction

The last decade has witnessed an increasing demand for meat that has been produced in a sustainable way, i.e., meat that provides a better balance between environmental, economic, and social (EES) aspects. The sustainability needs improving agricultural production systems which requires the promotion of farm practices that provide high-quality, affordable food in sufficient quantity while ensuring appropriate economic returns and minimizing negative environmental effects (Pelletier et al. 2008). Some countries, such as Brazil, have promoted the production of a wider variety of beef products to give consumers more choices. Essentially Brazil is producing two types of beef, i.e., organic and conventional. Conventional beef is produced either in specialized beef farm or mixed[1] crop-beef farm. Mixed crop-beef farming is an agricultural system in which a farmer conducts crops and livestock farming practice together. Mixed crop-beef developers can greatly improve the productivity of a beef system by improving carrying capacity and providing higher quality pasture.Organic beef is a type of farming in which it is not allowed to use synthetic inputs such as medicines, fertilizers, and genetically modified organisms (Chander et al. 2011). The aim of beef organic farming is to establish a method for avoiding environmental problems and promoting the quality and safety of beef (Nardone et al. 2004). Although currently mixed crop-beef and organic system cover a small share of the total beef production in Brazil, experts in beef production perceive these two systems as potential for future beef production in Brazil. However, there is limited knowledge regarding sustainability performance of the different beef production systems. A number of studies evaluated environmental aspects of beef production (Beauchemin et al. 2010; Casey and Holden 2006; Cederberg and Stadig 2003; Dick et al. 2014; Nguyen et al. 2010; Ogino et al. 2007; Pelletier et al. 2010; Röös et al. 2013; Roy et al. 2012; Stewart et al. 2009). Nevertheless, none of these studies focused on the three dimensions of sustainability simultaneously. Hence, an assessment of the EES performance of conventional and organic beef systems in Brazil should give insight into the sustainability consequences of these two production systems. Therefore, the objective of this study is to evaluate the EES performance of the conventional (based on specialized and mixed crop-beef farming) and organic beef production systems in Brazil.

2. Methods

2.1. Description of the systems

EES issues relevant for soybean production systems were listed in a study that performed by Pashaei Kamali et al. (2014). We limited the evaluation to some key EES issues for beef production due to data restriction,

[1] In this study mixed farm refers to On-farm mixing which enables the recycling of resources generated on a single farm.

methodological feasibility, and geographical relevancy of issues. Four EES issues were selected: Global warming, land occupation, primary energy use, profitability and animal welfare. Data for the conventional production systems were derived from Embrapa[2] cooperative data base which is aggregated farm data for two biomes in Brazil (Cerrado and Pampa) and represent average contemporary Brazilian conditions in these biomes. Comparable, broadly representative data were not available for mixed crop-beef and organic system in Brazil. Hypothetical mixed crop-beef and organic models were therefore designed to capable comparability with the conventional systems while reflecting key similarities and differences from conventional production technologies. The scenario model for the mixed crop-beef and organic beef were defined based on Brazilian expert opinion and broad trends identified through a literature review of comparative inputs in conventional (specialized and mixed crop-beef) and organic beef production systems. The simulated production unit used in this study for all systems consisted of a herd starting with originally 608 cows, 36 bulls.

2.1.1. Specialized beef system

The normal production system in Brazil is to produce animals under pasture condition. A small amount of animals (6.7% of slaughtered animals), in 2008 were fed in feedlots for a short period (Ferraz and Felício 2010; Somwaru and Valdes 2004). Therefore, in this study EES performance were calculated on a whole-herd basis in one year for grass-finishing systems, where the animals are able to continuously graze on the natural pasture throughout the year with little or no supplementation. Housing is not utilized in Brazilian beef production system, hence all manure is assumed to be deposited directly to pasture. The description of specialized beef system was based on representative beef production system in two biomes (Cerrado and Pampa). Main characteristics of specialized beef system are presented in (Table 1).

2.1.2. Scenario 1: Mixed crop-beef system

In the mixed crop-beef systems, livestock and crops are produced within a coordinated framework. In this system cattle graze in pasture-crop land, thus contributing to the relatively short beef production period. Crop–livestock systems that are spatially and temporally mixed can occur through various combinations of the following: (i) rotations of grain crops with perennial pastures; (ii) short rotations of grain crops with annual or short-season pastures; and (iii) utilization of grain crop residues for livestock grazing (Sulc and Tracy 2007). The mixed crop-beef model in this study was the third one. The crop was considered to be sold and assumed to be leguminous crops which increase the nitrogen of the soil and helps pasture improvement as well as reduce farm costs. The sold legumes were not included in this analysis, but the quantity of resources used for producing the crops were corrected in the computations. The herd evolution was performed using a method similar to that in the specialized beef system; with a starting point of same number of cattle. Main assumptions of mixed crop-beef system are presented in Table 1.

2.1.3. Scenario 2: Organic beef system

Organic animal husbandry is defined as a system of livestock production that promotes the use of organic and biodegradable inputs from the ecosystem in terms of animal nutrition, animal health, animal housing and breeding. It deliberately avoids the use of synthetic inputs such as drugs, feed additives and genetically engineered breeding inputs. In general, organic beef in Brazil comes from cattle raised in pastures for the majority of their lives. Unlike traditional or conventional systems of production, organic production systems are governed by a set of standards that must be strictly followed by producers. Main assumption and characteristics of organic system was presented in Table 1. There seem to be no fundamental differences between the specialized and organic cattle nutrition in Brazil since both of them are pasture base.

[2] Embrapa: Brazilian Enterprise for Agricultural Research

Table 1: Main characteristics assumed for different farming practices and basic assumption for each systems

	Farming practices				
Description	**Conventional beef**	**Mixed crop-beef**	**Reference/sources**	**Organic**	**Reference/sources**
Pasture (ha)[a]	1600	1600	Assumed similar to base scenario	1600	Assumed similar to base scenario
Composition pasture type	Native pasture	Native pasture with Leguminous	(Cederberg and Mattsson 2000; Dick et al. 2014)	Native pasture	
Phosphorous fertilizer (kg/ha/yr [c])	-	60[b]	(Embrapa soja, 2012)	-	-
Potassium fertilizer (kg/ha/yr)	-	76	(Embrapa soja, 2012)	-	-
Lime (kg/ha/yr)	-	400	(Embrapa soja, 2012)	-	-
Replacement rate (%)	20	12.5	(Dick et al. 2014)	20	Expert opinion
Age of first calving (month)	30	24	Expert opinion	32	Expert opinion
Calving rate	70	75	(Rearte and Pordomingo 2014)	70	(Rearte and Pordomingo 2014)
Stocking rate (AU/ha)[d]	1	1.6	(Oliveira et al. 2006)	0.8	(Oliveira et al. 2006)
Calf Mortality rate (%)	6	4	(Oliveira et al. 2006)	10	Expert opinion
Manure management	Pasture deposition	Pasture deposition	(Dick et al. 2014)	Pasture deposition	(Dick et al. 2014)
Total number of slaughtered cattle (yr)	731	796	-	640	-
Average age of slathering (months)	40	26	(Oliveira et al. 2006)	44	Expert opinion
Average weight of slaughter for female (kg)[e]	475	500	(Oliveira et al. 2006)& expert opinion	450	(Casey and Holden 2006)
Average weight of slaughter for male	475	520	Corrected based on (Dick et al. 2014)	472	(Casey and Holden 2006; Dick et al. 2014)
Number of labour (person.ha^{-1}.yr^{-1})	3	6[e]	Expert opinion	3	Expert opinion
Manure handling	Pasture	Pasture	(Dick et al. 2014)	Pasture	(Dick et al. 2014)
Crop farm in mixed crop-beef farm	-	Soybean	Expert opinion	-	-
Prices					
Selling price of cattle to slaughtered house (R$/@)[g]	90	90	90	90	(IBGE-SIDRA 2012)
Price premium (%)	-	-	-	30	Expert opinion
Salary of labour (R$/hr[h])	1.14	1.14	Embrapa	1.14	(IBGE 2012)

a. ha: hectare
b. Part p fertilizers were provided by manure
c. Yr: year
d. AU: Animal unit
e. Kg: Kilogram
f. Labour working for both cattle and crop farm
g. Selling price in farm gate is for 15 kg of cattle weight
h. R$: Brazilian Real
i. hr: hour

2.2. Environmental performance

2.2.1. Global warming

The environmental impact of beef production systems was evaluated by life cycle assessment (LCA). LCA is a method that evaluates the environmental impacts along the entire life cycle of a product (Guinée 2002). LCA relates the environmental impacts of the defined production system to the functional unit (FU) (Guinée et al., 2002), which is the main product of the analyzed system in quantitative terms, and defined here as one kilogram of live weight. The system boundary is cradle-to-farm gate (Figure 1). The animal population of systems was estimated from the simulation of herd evolution as recommended by (IPCC et al. 2006). The LCA began when the initial animals were weaned, continued through the meat production cycles, and ended when the initial cows and bulls were fully replaced (Dick et al. 2014). To estimate greenhouse gases (GHGs) in beef production in different systems, the following GHGs sources were included: (1) on-farm methane (CH_4) emissions from cattle and manure, (2) on-farm nitrous oxide (N_2O) emissions from manure and soils, (3) off-farm N_2O emissions from Nitrogen (N) leaching, run-off and volatilization (indirect N_2O emissions), (4) carbon dioxide (CO_2) emissions from on-Farm energy use (e.g., fuel and electricity). Animal housing is not common in Brazilian cattle system, so the energy use related to housing was not considered. Emission related to production of medicines and vaccines were excluded in this study due to lack of data.

To assess the impact of a production system on global warming (GW), we quantified emissions of CO_2, CH_4 and N_2O. Emission of CO_2, CH4, and N_2O were summed based on their equivalence factors in terms of CO_2-equivalents (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (IPCC et al. 2006). Emissions were subsequently summed based on their equivalence factor in terms of CO_2. For this study, impacts were calculated on a whole-herd basis for each class of cattle separately and per kg of live weight (LW) produced in each system. For the cow–calf system and grass-finishing systems modelled, housing is not utilized hence all manure is assumed to be deposited directly to pasture.

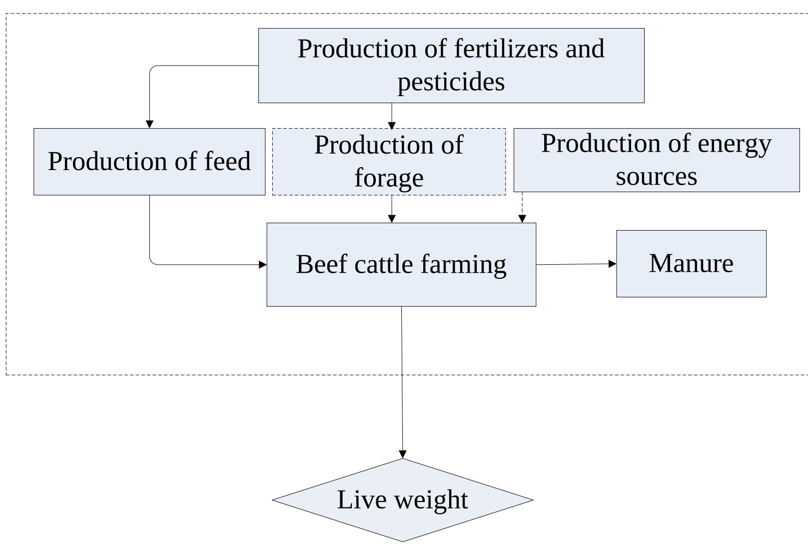

Figure 1: System boundary

Tier II method were applied for calculating enteric methane emissions due to the sensitivity of emissions to diet composition and the relative importance of CH_4 emissions to total GHG emissions in beef production (Pelletier et al. 2010). The enteric methane fermentation was quantified based on gross energy intake (GEI) of the cattle, which was calculated based on animal requirements and feed energy values. Daily net energy requirements for cattle in each stage of production were estimated from energy expenditures for maintenance, activity, growth, pregnancy, lactation and work as appropriate. The gross energy (GE) intake required to meet energy requirements was then estimated taking into account the energy density of the diet (Dick et al. 2014). Enteric CH4 emission was calculated from gross energy intake using the CH4 conversion factors (Ym) for each diet. Methane emissions from manure management were calculated following IPCC (2006) Tiers I. Tier I method was applied for manure management given the trivial methane emissions associated with solid manure management, which is common to all systems modelled (Pelletier et al. 2010). Nitrogen excretion estimates were used to calculate direct N_2O emission, ammonia and N_2O emissions from manure management and indirect N_2O emissions from nitrate leaching following (IPCC et al. 2006) emission factors.

The physical occupation of land areas was measured as the area (m^2. yr^{-1}) used for production of one kg of beef during one year. In calculations related to land occupation it was considered. For this study, we considered land for animal grazing (pasture). Primary energy use for producing beef was estimated based on total energy consumption in each phase. The energy required to produce beef included the energy used to produce farm inputs including transports (i.e., seed, fertilizers and agrochemicals) and energy use for field operation (fuel and electricity to operate agricultural field equipment). The energy required for buildings and agricultural machinery was ignored due to the lack of data and its small contribution to the total emissions (Pradhan et al. 2008). The primary energy use was calculated as the mass (kg) of material multiplied by its energy (MJ/kg).

2.2.2. Economic performance

To explore economic performance, profitability was evaluated. Operating profit was selected as an indicator for profitability, and was quantified as total revenue minus operating costs minus depreciation (Hillier et al. 2010). Operating costs are costs related to vaccination, medicines such as fertilizers, antibiotics, ear tags, fuel, electricity, repair, maintenance, operating interest, insurance, hired labor and transportation. Operating profit was quantified by revenue minus operating cost minus depreciation. Total operating costs is an indicator of the relative success of operations in terms of their ability to meet short-term financial obligations (McBride and Greene 2009).

2.2.3. Social performance

Animal Welfare focuses on beef cattle production including on-farm management of beef cattle. Animal welfare was evaluated based on main principles of welfare quality (assessment protocol for cattle). Four main principles are identified by this protocol: (P_1) good feeding, (P_2) good housing, (P_3) good health, and (P_4) appropriate behavior. The last one is out of the scope of this study due to lack of data. Regarding principle (1), (2), and (3), a number of welfare criteria were defined based on the aforementioned protocol for farm, transportation, and slaughtering. We defined eight welfare criteria for the farm level: (C_1) absence of prolonged hunger, (C_2) absence of prolonged thirst, (C_3) comfort around resting, (C_4) thermal comfort, (C_5) ease of movement, (C_6) absence of injuries, (C_7) absence of disease, and (C_8) absence of pain induced by management procedures (Welfare Quality 2009). The survey was carried out from April 2013 to January 2014 in Brazil. An invitation to complete an online questionnaire was distributed by email to the personal networks of SALSA[3] members. The questions were structured as closed-end questions and referred to eight welfare criteria for beef cattle farms. Respondents were asked to assign importance to each issue, using a five-point Likert scale, where 1 represented the "worst" or the poorest situation, 2 represented "worse" situation 3 represented "neutral" situation (neither bad nor good), 4 represented "good" situation and 5 represented the "best" situation (no more improvement is needed). Finally, an overall performance score is computed as the average over all indicators scores.

3. Results and discussion

Results from previous beef production studies showed large variability in terms of sustainability performance. The methods applied in these studies (e.g., system definition and characterization factors) were

[3] Knowledge-based Sustainable vAlue-added food chains: innovative tooLs for monitoring ethical, environmental and Socio-economic impActs and implementing EU-Latin America shared strategies.

different, which likely contributed to the variability in the results. However, the results of environmental evaluation of beef production obtained in this study fall within the range of values from previous studies. Total emission was low for specialized and mixed crop beef production compared to the organic and specialized beef production (Table 2). This conclusion is consistent with previous research, which has shown that higher quality diets and increased growth rates, reduce cattle CH_4 and manure N2O emissions, both of which are key contributors to life cycle emissions (Dick et al. 2014). Short days at grass and high productivity of the mixed crop-beef system are the main reasons for low emission. In the mixed crop beef production, crop residues (fibrous by-products) resulting from the cultivation of cereals and oil plants were the major source of nutrients for beef cattle and caused higher yields and better quality of the beef. The difference between the different beef production is due to the lower quality of the forage consumed by the animals in the specialized and organic compared with the mixed crop-beef and is based on the differences in dry matter (DM) intake/animal/day, the Ym, the digestibility, and the pasture use efficiency related to the time required to produce 1 kg live weight (Dick et al. 2014; Pelletier et al. 2010). Legumes in mixed crop-beef farms act as cover crops and green manures preventing soil degradation (Dick et al. 2014). Another benefit for incorporating legumes crops into pasture is the significant amount of available N added to the farm through dinitrogen fixation (Sulc and Tracy 2007).

Beef production in Brazil does not differ dramatically between organic and conventional systems. A few differences between these two systems were distinguished, for instance using antibiotics and medicines in conventional beef system which is not allowed in organic system. In this study organic beef had the highest GHGs emission since in this system cattle have the longest days at grass, lowest stocking rate, highest mortality rate and lightest animals sold for slaughtering. The aforementioned factors in organic system cause slightly higher GHGs emission compared to conventional system. Although this result has contradiction with other studies (Casey and Holden 2006), however, the difference easily can be explained by differences in feedlot or grain based finishing versus pasture based finishing (this study). Organic beef usually produced for quality rather than weight (Casey and Holden 2006), therefore the moderately high GHGs emission arises because of the lower weight per animal.

The land occupation for organic farm was higher (53.1) compared to conventional system (Table 2). Higher land occupation in organic system can be explained by a lower total production of slaughtered cattle (due to high mortality rate and high grazing period). Organic beef production system is commonly associated with lower productivity in comparison with conventional production system (Casey and Holden 2006; Nardone et al. 2004). This study also confirmed that there is difference between organic and conventional beef system productivity. Therefore, the main challenge for organic production system to improve overall sustainability is to increase productivity without negative impact on the environment (Nardone et al. 2004). A number of research experiments have shown that under carefully controlled management and better performance conditions organic production system has the potential to achieve comparable output with those in conventional one. The lowest land occupation was in mixed crop-beef farm (38.6) due to high productivity in this farming practice (Dick et al. 2014). Regarding energy specialized and organic beef use less energy compared to specialized crop-beef beef production (Table 2). The result of land occupation in this study is in similar to the studies performed by Pelletier et al. (2010) for intensive pasture finishing beef cattle. For specialized and organic beef production there was not any energy related to feed production, fertilizer production and transportation of feed and fertilizers. However, for mixed crop-beef system due to having crop production in the farm part of energy use was allocated to beef production system. Indeed energy use in this study is much less than the other studies which had feedlot or grain based finishing system (Nguyen et al. 2010; Pelletier et al. 2010). Hence, we could compare this study only with limited number of studies which had pasture based systems (Cederberg et al. 2009).

Despite the interest in organic beef production due to some environmental and human safety reasons, there is a little information concerning the relative costs and returns of organic beef production. Although our study showed that organic production has higher production cost compared to the conventional beef production, however, the operating profit per kg live weight was highest for organic beef, followed by mixed crop beef, and specialized beef. Without organic price premiums, the average annual profits of the conventional production system were higher than the organic production system. (Azadi and Ho 2010; Fernandez and Woodward 1999). Higher cost of organic beef is related to veterinary cost, certification cost, long staying of animal in farm and low productivity. Mixed crop-beef systems is resulted in higher livestock carrying capacity and more consistent farm profitability compared with conventional system (Sulc and Tracy 2007). Lowering input levels increase productivity, and maintaining or increasing profitability. Crop residues represent a vast feed resource available to livestock producers that can effectively reduce feed costs. According to (Sulc and Tracy 2007) in southern Brazil, research and experiences on commercial farms have demonstrated that mixed crop–livestock grazing systems can improve net returns eightfold over the traditional extensive stocker grazing systems and 1.5-fold over soybean grain production systems. Grazing winter cover crops (such as soybean) did not reduce subsequent grain yield when animal stocking was managed (Sulc and Tracy 2007). Animal welfare score of the organic system was slightly higher (4.5) than the score of the other farms. Animal welfare is a basic principle of organic production, and organic beef cattle farmers and managers have a more explicit responsibility for the health, welfare, and treatment of the animals. Organic farms are obliged to obey certifications and standards, which are proposed for organic beef production. The mixed crop beef system had a slightly lower average score (4.2) than the organic farm. The specialized beef system had the lowest animal welfare score (3.98).

Table 2: Environmental and economic performance of beef cattle production in different farming practice

Impact	Unit	Conventional beef	Mixed crop-cattle beef	Organic beef
Global Warming potential	kg CO2 eq.	16	14.2	15.7
Primary energy use	MJ eq.	13	12	13.7
Land occupation	m^2 x year	41.4	38.6	53.1
Operating profit	US $	2.91	3.1	4.3

The management of small ruminant organic farming is fundamentally based on the choice of an appropriate forage system and on good knowledge of climatic and animal production system. Furthermore, prices of organic beef might have great variability, immature nature of the organic market (McBride and Greene 2009). Organic beef cattle productivity are is subject to disease, weather and other factors, which mean that the output stay rather volatile also in the future and may present particular risks and ways of managing risks (Van Bueren et al. 2002). Additional risks of producing organic beef, such as production, marketing and policy risks may cause differences in farmers' risk attitudes. For a risk-neutral farmer it is optimal to produce organic beef; however, for a risk-averse farmer it is only optimal to produce organic products if policy incentives are applied, or if the market for the organic beef becomes more stable.

4. Conclusion

The potential environment impacts of 1 kg live weigh beef produced in different beef production systems in Brazil were shown to differ considerably. (Hanson et al. 2004). Different features of systems contribute to EES performance of each system. The production period of the beef and the quality and production of the pastures determine the GWP, land occupation, energy use and profitability. Concerning product quality, there is little evidence for a system-related effect on product quality due to the production method. It is concluded that the benefits of the basic standards are primarily related to environmentally friendly production and to the animal welfare issue while the issues of animal health and product quality are more influenced by the specific farm management than by the production method. There is evidence to support the assumption that organic livestock farming creates stronger demands on the qualification of the farm management, including the higher risk of failure. As a consequence, quality assurance programs should be established to ensure that the high demands of the consumers are fulfilled. (Sundrum 2001). Alterations in diet composition and animal husbandry practices in mixed crop-beef and organic systems have been proposed as a means of reducing negative environmental impacts and improving economic performance of beef farms (Beauchemin et al. 2008; Eckard et al. 2010; Johnson and Johnson 1995; Martin et al. 2010). We hypothesize such systems will be economically competitive and less environmentally harmful than the specialized beef system. However, for this to become reality it needs to invest in research and training for establishment of management systems adapted to environment and sociological context. Moreover, crop–beef farming systems by producing of different products can provide additional marketing opportunities beyond the conventional commodity markets

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Comparing Alternative Nutritional Functional Units for Expressing Life Cycle Greenhouse Gas Emissions in Food Production Systems

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ABSTRACT
Life cycle assessments (LCAs) of crop production systems and food products commonly report results using functional units of mass. While these functional units appear to facilitate comparison between different products, they fail to account for substantial differences in a multitude of benefits humans derive from food products, especially nutritional value. This study explores the effects of using different functional units, including mass, serving size, energy content, protein content and a composite nutrient score, on four food's life cycle global warming potential (GWP). Process-based LCA models of almonds, processed tomatoes (diced and paste), and rice, all produced in California, are used to calculate GWP. The results show that rice has the highest GWP except for the energy-content based functional unit, and almonds the lowest GWP except for the mass-based functional unit, and the performance of all the products are significantly affected by the choice of functional unit. The composite nutrient score functional unit appears to magnify the differences among the foods because it accentuates nutrient density, and thus foods like almonds perform better and foods like rice perform worse. Transparency in reporting and reporting multiple functional units is recommended for future studies.

Keywords: carbon footprint, nutrition, rice, almond, tomato

1. Introduction

Life cycle assessments (LCAs) of crop production systems and food products commonly report results using functional units (FUs) of mass of the harvested crop or food product. While these FUs appear to facilitate comparison between products, they fail to account for substantial differences in how various food products are actually used by consumers and the types of benefits consumers derive from these foods, especially nutritional value (Heller et al. 2013). Though meal- and diet-based FUs address this shortcoming (Carlsson-Kanyama 1998), they include functions beyond nutrition, such as those derived from culture. In many cases these additional functions improve the quality and interpretation of analysis and provide important context for decision-making; however, meal and diet-based LCAs may limit generalizability and comparability of results. This study explores a variety of FUs for diverse food products, including mass, serving size, energy content, protein, and a composite nutrient score, and examines the implications for how these products are perceived in terms of their greenhouse gas (GHG) footprints as calculated using LCA methods.

1.1. Previous work

The issue of FU selection in food LCAs has been the subject of discussion in diet-level assessments, meal-based assessments, comparison of organic and conventional production methods, and comparisons across food products or product categories, for example Camillis et al. (2012), Carlsson-Kanyama (1998), Davis and Sonneson (2008), Davis et al. (2010), Dutilh and Kramer (2000), Heller et al. (2013), Notarnicola et al. (2012), Saarinen (2012), and Van Kernebeek et al. (2013), to name but a few. The primary issue is selecting functional units that reflect both the goal of a study and the role of a particular food product in a diet.

While many comparative studies reject mass-based functional units, the European Food Sustainable Consumption and Production Round Table Working Group 1 recommends that business to consumer reporting use a functional unit of 100 g or ml (Camillis et al. 2012), illustrating the continued debate and ambiguity in the appropriate FU for food LCA studies. Complicating the issue of mass-based (or volume based) functional units and nutrition-based functional units is the fact that actual diets are not necessarily healthy, and may over-consume particular nutrients, complicating the selection of a FU when the role of a food or nutrient in a diet or meal is part of the functional unit. For example, Van Kerbeck et al. (2013) examined composite nutritional indices for use as FUs. They found that nutritional indices based on diets with and without protein intake caps arrived at different results. This and other studies highlight the challenge of developing FUs for food LCAs that are both based on scientifically relevant metrics of a food's value to the human diet, and reflect the real-world role of a food in the human diet.

Other issues that affect LCA-based food comparisons are those that are more universal to comparative LCAs and meta-analyses of LCAs, such as incommensurate system boundaries, differing key assumptions and a lack of transparency in reporting that prevents reinterpretation of results for new or alternate functional units.

1.2 Study Approach

This study compares three dissimilar foods produced and consumed in California; almonds, rice, and processed tomatoes. These foods were selected for three reasons: first, LCAs were conducted for these three products by the same research team leading to adequately similar system boundaries, data sources, and modeling approaches; second, because each food has significantly different roles in the human diet, they provided a good test bed for comparing different functional units; third, each of these foods is important globally due to the scale of production and their share of global or national production. California almond production constitutes 83% of global commercial production (USDA Office of Global Analysis 2013), California produces 96% of US processing tomatoes and is the single largest global producer (Economic Research Service 2012), and California is the second largest state in the US for rice production at approximately 2 million metric tons per year, 40% of which is exported to the international market (California Rice Commission 2014).

2. Methods

This study uses process-based LCA models of whole almonds (Kendall et al. 2012), processed tomatoes (Brodt et al. 2013), and rice, all produced in California. Because the rice LCA model was restricted GHG emissions, the only impact assessment category included in this study is GWP_{100} (Intergovernmental Panel on Climate Change 2007). Models were developed to cover all typical California production inputs, equipment operations, and yields on an annual basis, based on University of California Cost of Production studies and consultation with crop experts. Similar methods and system boundaries were used for all three products. In order to preclude confounding the results with different packaging options for different products, all results were calculated for the final products, to the food processing facility gate, without including any packaging. The almond study reports results for raw brown skin almonds. The processing tomato study evaluates two processed tomato products, canned tomato paste and canned diced tomatoes. The rice study reports a weighted average of white rice (90%) and brown rice (10%) based on their estimated production rates because these two products require different levels of milling.

The GWP results for these products are reported here in units of kg CO_2-equivalent per kg food product (kg CO_2e/kg):

- Whole almonds: 0.51 kg CO_2e/kg almond
- Tomato paste: 0.85 kg CO_2e/kg tomato paste
- Diced tomatoes: 0.17 kg CO2e/kg diced tomatoes
- Rice: 1.7 kg CO_2e/kg rice (90% white, 10% brown)

The functional units considered in this study include mass, serving size, calories, protein content, and a composite nutrient score. The recommended serving size of each product is based on common manufacturers' nutrition facts labels on commercially available products and industry standards (Almond Board of California). The following serving sizes are used in this study: rice, 45 g; almonds, 28g; tomato paste, 33 g; and diced tomatoes, 122 g.

The calorie content and protein content estimates for each food product are based on the U.S. Department of Agriculture's most recent database on nutrients in food (U.S. Department of Agriculture Agricultural Research Service 2013). Table 1 report the nutrient values assigned to each food product.

The composite nutrient score used in this analysis is drawn from Arsenault et al. (2012). Arsenault et al. used a regression analysis on real U.S. diets relative to a "Healthy Eating Index," or HEI. By doing so, the authors identified eight nutrient characteristics that were significant in determining the nutritional value of foods. Five of these characteristics contribute positively to nutrition; protein, fiber, calcium, unsaturated fat, and vitamin C. Three characteristics contribute negatively; saturated fat, added sugar, and sodium. Table 1 includes these eight nutrients for the food products considered in this study, as determined by the U.S. Department of Agriculture's Agricultural Research Service (2013).

The regression analysis results in the following algorithm for determining a weighted nutrient density score (WNDS):

$$WNDS = 100 \times (1.4 \times \mathrm{g}\frac{\mathrm{protein}}{50} + 3.3 \times \mathrm{g}\frac{\mathrm{fiber}}{25} + \mu\mathrm{g\ calcium} + 2.51 \times \mathrm{g\ unsaturated}\frac{\mathrm{fat}}{44} + 0.37 \times \mathrm{mg\ vitamin}\frac{\mathrm{C}}{60} - 2.95 \times \mathrm{g\ saturated}\frac{\mathrm{fat}}{20} - 0.52 \times \mathrm{g\ added}\frac{\mathrm{sugar}}{50} - 1.34 \times \mathrm{mg}\frac{\mathrm{sodium}}{2400}) \qquad \text{Eq. 1}$$

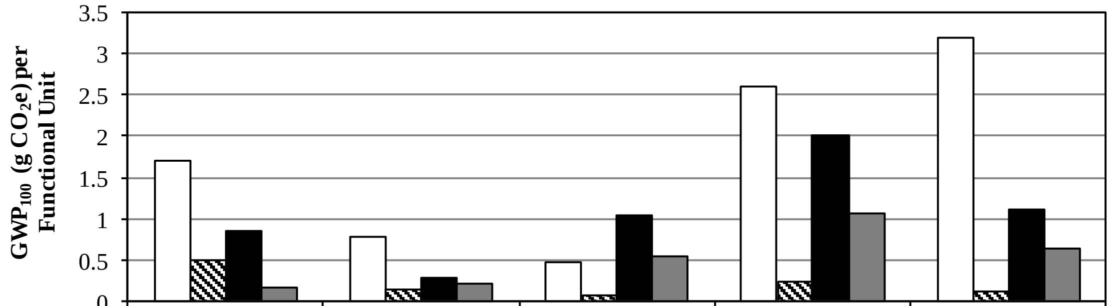
Table 1. Food product nutritional characteristics.

Characteristic	Unit	white medium and short grain rice, un-cooked	brown rice medium & short grain, un-cooked	Weighted average of 90% white, 10% brown rice	tomato paste	tomatoes, crushed canned	almonds, raw
g protein	per 100 grams	6.5	7.5	6.6	4.3	1.6	21
g fiber		2.8	3.4	2.9	4.1	1.9	13
mg calcium		3.0	33	6.0	36	34	270
g unsaturated fat		0.20	1.9	0.37	0.23	0.16	44
mg vitamin C		0	0	0	22	9.2	0
g saturated fat		0.14	0.54	0.18	0.1	0.040	3.8
g added sugar		0	0	0	0	0	0
mg sodium		1.0	4.0	1.3	59	19	1.0
kcal (food calorie)		360	360	360	82	32	580
g product	per 100 kcal	28	28	28	120	310	17
g protein		1.8	2.1	1.8	5.3	5.1	3.7
WNDS	na	15	19	15	94	86	75

3. Results

Greenhouse gas emissions, reported in units of kg CO_2e, vary widely for the four different food products, and also vary substantially within each food product depending on which functional unit is used (Figure 1). The WNDS-based functional unit is calculated using Equation 1 based on the mass of product needed for 100 kcal.

In four of the five FU cases rice performs worst, raw almonds perform best in all cases except the mass-based functional unit, and canned tomatoes perform best on a mass-basis and second best in all other cases.



	Per kg	Per Serving (×10)	Per 1000 kcal	Per 100 g protein	Per point of WNDS in 100 kcal
Rice (uncooked)	1.7	0.78	0.48	2.6	3.2
Raw almonds	0.51	0.14	0.086	0.24	0.12
Canned tomato paste	0.85	0.28	1.04	2	1.1
Canned diced tomatoes	0.17	0.21	0.54	1.06	0.64

Figure 1. GWP_{100} for four California food products using five functional units; mass, serving size, energy content, protein content, and a composite nutrient score.

4. Discussion

Results demonstrate how widely environmental impacts can vary, depending on the nutritional characteristic of interest. Due to high methane emissions from flooded field production, rice tends to have higher GHG emissions than the other three foods for all functional units, except for kilocalories. With its high carbohydrate content, rice delivers many more kilocalories than vegetables in any form, even the highly concentrated tomato paste, and therefore the environmental impacts are spread across more kilocalories. However, since white rice tends to be low in other nutrients, its impact on the basis of the WNDS is much higher compared to the other foods. Almonds perform better than rice on the basis of kilocalories, due to their high fat content. They also have an order of magnitude lower impact when assessed on the basis of protein, and even lower impacts when assessed according to their WNDS because of their high nutrient density.

Tomato paste offers an interesting case due to energy-intensive processing required to produce this concentrated product. While the intensive processing results in a relatively high emissions profile on a mass basis, it also serves to concentrate nutrients, balancing the higher emissions with a higher nutritional profile. For this reason, its GHG emissions profile per point of WNDS, while still higher than that for diced tomatoes and almonds, is not as distant from those two products compared to its profile on a mass, kilocalorie or protein basis. This analysis using the WNDS suggests that diets that are more heavily dependent on low-nutrient foods, such as rice, for functional nutrition, might incur higher environmental costs (as well as overconsumption of carbohydrates) than those that rely more on nutritionally-dense foods, assuming maintenance of a similar total nutrient content across diets.

Another observation that may be drawn from Figure 1 is that the WNDS seems to magnify differences among the products. In particular, the impact from rice looks significantly worse, while that from almonds looks much smaller, compared to the other foods. This is mostly due to differences in nutrient density of the two foods, but highlights the importance of transparent reporting when a composite nutrient score is used as the functional unit. Moreover, the WNDS, in this analysis, serves primarily as a mechanism for representing the composite nutritional density of a food based on a limited number of beneficial nutrients. Since all four of the food products are unprocessed or minimally processed foods that are generally considered healthy components of a diet, their WNDS are not highly affected by subtractions (in Equation 1) due to unhealthy factors, namely saturated fat, added sugar, and sodium. Evaluating the impact of these four foods when incorporated as ingredients in more processed products (e.g. almonds in a candy bar or tomato paste on a pizza) that have more of these unhealthy components, would present a more nuanced and complex use of the WNDS, and would more likely reflect the majority of actual consumption of these foods, than in the analysis here. In addition, changing science, subjective or culturally mediated notions of health, and the role of a particular food in a 'balanced' diet (meaning that almonds and rice are not likely to be substitutable in a diet), means that composite indicators such as the WNDS are likely to change over time.

It should be noted that the relative rankings of the four foods are also highly influenced by the environmental impact being assessed, which in this case is GHG emissions. This impact puts rice at a large disadvantage due to the peculiarities of its production system and its high methane emissions. If assessed on other impacts, such as other pollutant emissions or wildlife habitat provision, rice might perform as well as, or better than, the other foods, even with its lower vitamin and protein content. This is a well-understood shortcoming of carbon footprints, or any environmental assessment that focuses on a single impact category.

While researchers may be able to interpret or reinterpret FUs in a way that is meaningful to them, selecting a FU for LCAs targeting consumers is more challenging. A per-serving FU may make the most intuitive sense to a consumer, assuming the serving size reflects typical consumption. However, serving size does not capture the whole-diet or whole-meal perspective which may be important if environmental and dietary decisions are to be linked in a meaningful way in consumer labels. One lesson from our analysis is that presenting results using more than one FU might be desirable, in order to accurately convey the costs of different dietary values that might be of interest to different consumers. However, this approach also risks confusing consumers, if too many FUs show too many conflicting results.

5. Conclusion

Environmental and nutritional policies tend to be formulated in isolation from one another, sometimes resulting in sub-optimal recommendations, such as the advice to consume more fish from fisheries on the verge of ecological collapse. By combining LCA with nutritionally focused functional units, we can begin to bridge this gap between nutritional and environmental sciences and understand the true environmental costs of different nutritional profiles.

Though a recommendation on the FU for any particular study may be driven by a study's goal, audience, scientific developments (particularly with regard to nutrition and diet), or cultural or regional dietary differences, one recommendation applicable to all studies can be made: all LCAs of food products should report results in such a way that alternative functional units can be calculated by other researchers. Thus, a study using a FU based on a composite nutrient indicator should also provide results based on a reference flow such as mass as well as information on nutrient characteristics of the food. As indicated in the discussion section, determining the appropriate FU for consumer-oriented studies is more challenging. There may be ways for interested consumers to use diet calculators, or link shopping lists to LCA information to better understand their diet's impact on the environment. However, the choice of a FU for a consumer-oriented labeling scheme is still unresolved.

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Life Cycle Assessment of Cheese Manufacturing in the United States

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ABSTRACT

A farm-gate-to-customer-gate life cycle assessment was conducted to evaluate the environmental and ecological impacts associated with US cheddar and mozzarella cheese manufacturing. Data collected from 16 cheese processing plants (ten cheddar plants and six mozzarella plants) were used to construct a life cycle inventory model, from raw milk delivery to the plant's refrigerated storage silo through delivery of packaged cheese and whey to the first customer. Our baseline allocation approach for energy and other resource use was the facility-supplied fractional estimation of co-products. Where specific information was not provided, revenue-based allocation was used for remaining inputs and emissions. The major environmental impact contributors to climate change were electricity usage (28.3% of total) followed by cheese transport and natural gas use representing 22.5% and 17.3%, respectively. Accumulated reductions in electricity consumption across the life cycle will have significant sustainability benefits. Additional metrics and normalization of the results are discussed in the document.

Keywords: Life cycle assessment, environmental impact, greenhouse gas (GHG) emission, energy use, cheese manufacturing

1. Introduction

Cheese is one of the highly recommended dairy products to consume as part of a healthy and balanced diet. However, cheese manufacturing is a complex process where a myriad of operations take place which are energy intensive. Therefore, it is necessary to estimate the environmental consequences associated with cheese manufacturing to make improvements that promote sustainability (Berlin 2002). In 2009, about 4.6 million tons of cheese products at more than 450 processing plants were produced in the United States, and there has been a trend toward increasing consumption of cheese with an average annual growth rate of 2.5% over the last decade (IDFA 2010). The production of cheese requires numerous resource inputs and outputs that contribute to environmental and ecological risks including greenhouse gas (GHG) emissions. This study provides information that will help the cheese industry to engage more sustainable practices, and reduce environmental consequences, while validating those reductions through science-based analysis. This work will also assist the cheese industry in positioning its products in the market place in terms of their sustainable attributes and be proactive with their own environmental initiatives and toward consumer concerns.

The study reported here was a part of larger effort to evaluate the cradle-to-grave life cycle assessment (LCA) of the environmental impacts associated with cheese consumption in the United States (Kim et al. 2013). This paper presents a farm-gate-to-customer-gate environmental impact analysis of cheddar and mozzarella cheese processing plants. This study focused on quantifying emissions to air, water and soil, cumulative energy demand, consumption of water and other natural resources, and assessing the impacts of inventory exchanges on climate change, resource depletion, human and ecosystem health. In particular, nine environmental impact categories associated with processing, packaging, and distribution in the production and delivery of a ton of packaged cheddar and mozzarella cheese were investigated. Cheddar and mozzarella cheese were chosen on the basis that they represent about 64% of all natural cheese produced and 80% of sales basis in the United States (IDFA 2010). The overarching goal of this work was to equip cheese industry stakeholders with timely, science-based information to further allow incorporation of environmental performance into decision-making and drive innovation.

2. Methods

2.1. Goal and scope

The main goal was to perform an analysis of potential environmental impacts associated with manufacturing of cheddar and mozzarella cheese and to provide cheese manufacturers with information to benchmark their performance against a 2009 industry average in the United States. The identification of the operations which have a major environmental impact will make possible to establish improvement actions and to quantify the impact reductions achieved. The scope of this work was a farm-gate-to-customer-gate impact assessment of a typical cheese processing plant. In particular, the specific operations included transport of raw milk to the plant, cheese and whey manufacture, and delivery of cheese and whey products to the first customer. The first customer can be a retailer for cheese and whey products, animal farm for whey as a feed, or other facility and country for further processing (Some whey is exported to China). This analysis was performed in compliance with ISO 14040 and 14044 standards for life cycle assessment (ISO 2006a; ISO 2006b).

2.2. Functional unit and system boundaries

The functional unit was defined as one ton (1,000 kg, dry-weight basis) of packaged cheddar and mozzarella cheese delivered to the plant's first customers. The system boundaries begin with the raw milk loaded into truck at the farm and end with delivery of packaged cheese and whey to the first customer via the plant's distribution truck. Landfill disposal of packaging is included in the model. In determining whether to expend project resources to collect data for the inclusion of specific inputs, a cut off criterion was established as 1% threshold for mass and energy. Exceptions to this exclusion were made in cases where significant environmental impact is associated with a small mass input. Also, if the data was readily available the elementary flow was not excluded simply because it was small. Even though the study is intended to be comprehensive in consideration of impacts resulting from cheese production, it is not a detailed engineering analysis of specific unit operations within each manufacturing facility. Thus, for example, we did not assign a specific energy requirement for the cheese making vat, cleaning-in-place (CIP), or starter culture operations; rather we used the information available at the manufacturing plant-scale, coupled with allocation of burdens to multiple plant products, to define the burden assigned to primary cheese, other cheese, dry whey, wet whey and other co-products. For this reason, it is important to state that all operations, as well as facility overhead (heating, lights, computers, etc.) are accounted in this study.

2.3. Gate-to-gate inventory data

Data collected specifically for this project were primarily derived from a survey created for cheese manufacturing plants. During 2010, processing companies that participated voluntarily were asked to complete a spreadsheet-based data entry template. Ten cheddar manufacturing facilities (0.55 million tons of cumulative production; 38% of US annual production), six mozzarella manufacturing facilities (0.35 million tons of cumulative production; 24% of US annual production) and one whey manufacturing facility responded. Collected data were based on calendar year 2009. A variety of plant sizes are represented with production ranging from 14 million kilograms per year to around 140 million kilograms per year. The average production of the 10 cheddar and 6 mozzarella plants from the survey was just less than 54 million kilograms per year of cheese, for both cheese varieties. The survey requested facility level data regarding purchases (energy, materials, chemicals, water, and truck fleet fuel), production (cheese and other products), and emissions (solid and liquid waste streams). The information was at the most refined level known by plant managers; however, in many cases only whole facility-level data were available. For example, most plants only reported a single annual electrical energy use. Plants were requested to provide an estimate for separate material and/or energy exchanges (inputs and outputs) associated solely for either cheese or whey products; this information was used to refine the allocation of material and energy exchanges between the co-products of cheese and whey on an individual flow basis. Whey processing can require more thermal energy and resource intensity per ton of finished product than cheese, depending on the degree of solids concentration. Additionally, since facilities produced other cheeses (not cheddar or mozzarella), wet whey, dry whey, other co-products (cream, butter, permeate, lactose, etc.), or a combination of these products, the delineation of the multiple co-products is important in the methodology. For the functional units chosen in this study, packaging is a re-sealable low-density polyethylene plastic bag. In addition to the primary product packaging, we collected survey information regarding secondary packaging (e.g., corrugated cardboard boxes, stretch wrap, paper bags, slip sheets and pallets). The survey also included information for transportation of raw milk to the processor, as well as distribution of final products to storage and first customer. The baseline vehicle was considered to be an insulated tanker truck for raw milk transport and a refrigerated truck for cheese or other co-products distribution. Empty return miles were also accounted. SimaPro© 7.3 (PRé Consultants, The Netherlands 2012) was used as the primary modeling software; the ecoinvent database, modified to account for US electricity (use of the average US grid excludes the variation in the results by region, EarthShift 2012), provided information on the 'upstream' burdens associated with materials like primary fuels and plant chemicals.

2.4. Life cycle impact assessment

We adopted the ReCiPe (Goedkoop et al. 2009) and the USEtox (Rosenbaum et al. 2008) methods with an additional inventory category, cumulative energy demand (Hischier et al. 2010). The two inventory categories, non-renewable fossil fuel (cumulative energy demand) and water depletion (ReCiPe inventory indicator), and the seven impact categories: climate change (IPCC GWP 100a), freshwater and marine eutrophication (ReCiPe midpoint), photochemical oxidant formation (ReCiPe midpoint), ecosystems (ReCiPe endpoint), human health (USEtox), and ecotoxicity (USEtox) are presented quantitatively. ISO 14044 standards do not permit combining multiple metrics into a single score, therefore the results of this study were reported as individual metrics for each of the impact categories.

2.5. Allocation

There are five potential co-products in this work. Because a variety of whey handling technologies are commonly used in the industry, allocation decisions are critically important. Some plants have virtually no processing of the whey after it is removed from the cheese vat, other plants have major processing of the whey making various protein concentrations or dried powders. Data is most commonly available only at the whole plant level. Therefore, to properly account for the energy and impacts that should be allocated to multiple products, a careful accounting of the differences in whey processing is included.

Some of the unit processes are unique to individual products and others are common to all other co-products. For all processes that can be clearly defined as being associated with a single product was assigned to that product. In the survey of manufacturing facilities, we asked for engineering estimates of the allocation of common inputs and emissions to internal facility operations and co-products. In addition, we collected the distribution of gross revenue generated by all the co-products from the facility. For inputs and emissions, these estimated allocation fractions were used as our principal source of information. For energy and other resource use in the plant, we used the plant-supplied engineering allocation. If the plant did not provide engineering estimated allocation fraction, we used an allocation based on the reported fraction of plant revenue assigned to that product. The burdens from milk production and transport were allocated based on the distribution of milk solids using the Van Slyke equation (van Slyke and Price 1979) which estimates the theoretical yield of cheese and whey products based on the incoming raw milk fat and protein content.

3. Results

3.1. Environmental impact assessment results

The inventory information collected from the surveys was converted into lifecycle inventory datasets, coupled with ecoinvent datasets and analyzed using SimaPro 7.3 software. Because the units of measure for each impact category are dissimilar, we report results as a contribution analysis by production inputs. Processing-related operation impacts were grouped into six categories: electricity, natural gas, fuel oil, chemicals, water and wastewater treatment. Packaging-related impacts were grouped into five categories: corrugated board, plastics, pallet, plywood and disposal. Distribution-related impacts were grouped into three categories: cheese transport, raw milk transport and other transport. Other transport here includes the transport of purchased chemicals, packaging materials, cream, and ingredients such as starter media, salt and rennet, etc. On cheddar cheese analysis, electricity usage contributes the largest impacts to climate change with 46.6% on processing stage followed by natural gas and chemical usage with 28.4% and 10.8 %, respectively. Packaging stage does not have significant environmental impacts representing only 7.7% contribution overall to climate change. On distribution stage, cheese transport stands for the largest contributor across all nine impact categories making up about 71.5% contribution followed by raw milk transport, 19.4% contribution to climate change. On overall interpretation from farm-gate-to-customer-gate, electricity usage contributes the largest impacts to climate change with 28.3% followed by cheese transport and natural gas use with 22.5% and 17.3 %, respectively. Similar results as climate change are observed on cumulative energy demand, thus these are not explained here. The freshwater depletion is dominated by processing stage, due to process water usage mostly for cleaning-in-place (CIP), representing 90.8%. The eutrophication impacts are affected by on-site wastewater treatment (WWT). The unit process for whey digestion was adapted from the ecoinvent dataset. This unit process is based on modern technology from Switzerland, and should be generally applicable in the US; however, the incineration of sludge is included in the original dataset, and this is not a common practice in the US. Thus, exchanges in the dataset which were identified as deriving from sludge incineration were removed. In addition, the assumed phosphorus loading to the treatment facility, for Swiss conditions, was 250 mg/L. Based on the survey and literature (Danalewich et al. 1998) reports, we reduced the WWT influent total phosphorus to 70 mg/L. The survey data were highly variable with some reporting post-treatment concentrations. Furthermore, not all plants reported P loadings so the average estimated load was used for all facilities. Freshwater eutrophication is driven by phosphorus loading, and for the impact assessment framework chosen for this study, there was no differentiation between phosphorus emitted directly into receiving waters and phosphorus applied to land with sludge. For marine eutrophication, which is driven by nitrogen emissions, we did not have any information in the survey, and therefore adopted the ecoinvent WWT dataset influent loading and emissions profile. Nitrogen compounds which are emitted are generally in a soluble form, and due to the prevalence of phosphorus limitations in freshwater, an assumption that all N emitted ultimately reaches marine waters is made in the impact methodologies. Because the WWT process is the dominant source of eutrophication, and the uncertainties associated with the inventory and impact methodologies, further evaluation of the damages actually resulting from WWT is needed. This analysis shows that there is a potential impact associated with these emissions, but does not demonstrate the fact of this damage to the environment. The photochemical oxidant formation impact is dominated by distribution stage representing 65.5% followed by processing stage which makes up 28.8% of overall. The primary driver to this impact is observed to be nitrogen oxides (NOx) emissions which are stem from truck tailpipe. Volatile organic compounds emissions from combustion associated with transport and manufacturing are also noticeable drivers. The ecosystems damage is also affected most by processing stage with 55.8% of overall. The primary driver to this impact is carbon dioxide emissions associated with electricity generation and heating fuel combustion followed by forestry for corrugated board packaging. The human toxicity and ecotoxicity impacts show similar result pattern dominated by processing stage which represents 77.1% and 66.0% of overall, respectively. The largest contributor to these impacts is heavy metal emissions to both air and water primarily driven from coal mining tailings and coal ash disposal in the electricity supply chain. However, these results should be interpreted with care because USEtox characterization factors for metals are highly uncertain according to Rosenbaum et al. (2008). A discussion of major contributors to each impact category is provided in Table 1 which is applicable for both cheddar and mozzarella cheese.

Table 1. Gate-to-gate drivers across environmental impact categories for both cheddar and mozzarella cheese.

Damage category	Major gate-to-gate impact drivers
Climate change	CO_2-equivalent emissions from fossil fuels combustion related to electricity generation, on-site heating fuels usage and diesel usage
Cumulative energy demand	Energy demand by electricity generation, natural gas usage and diesel fuel usage
Freshwater depletion	Process water usage
Marine eutrophication	Nitrogen compounds released into water from wastewater treatment; likely associated with nitric acid used in CIP operations
Photochemical oxidant formation	NOx and VOCs emissions from combustion associated with transport and manufacturing
Freshwater eutrophication	Phosphorus emissions associated with wastewater treatment with smaller impacts associated with manufacturing from coal mining for electricity generation. Phosphorus is likely emitted during CIP operations where milk protein residues are removed from piping and equipment
Ecosystems	Majority from CO_2 emissions associated with electricity generation, heating fuel and transport combustion; secondary from forestry for corrugated board packaging.
Human toxicity	Arsenic emission to water and heavy metals emissions to both air and water primarily from coal mining tailings and coal ash disposal in the electricity supply chain
Ecotoxicity	Chromium emission to water and heavy metals emissions to both air and water primarily from coal mining tailings and coal ash disposal in the electricity supply chain

3.2. Cleaning-in-place (CIP)

The cheese manufacturing industry has been active for many years in addressing the important issues associated with CIP technologies and chemical usage. As described in the previous section, CIP is a contributor to several impact categories. However, one of the limitations of the data available for this study was that only whole-plant information and therefore detailed analysis of the CIP was not possible. Specifically, not all chemical purchases, or energy consumption at the whole plant level could be attributed to CIP, thus the emphasis of manufacturers on managing CIP impacts remains important. Sustainability will be achieved through continual incremental improvement, and these improvements across many plants will cumulatively have a significant and measurable impact at the sector level. The granularity of data in this study highlights the need for further investigation at the plant level.

3.3. Normalization

We conducted a normalization test using the Impact 2002+ US midpoint assessment framework which was published by Lautier et al. (2010). The result for cheddar indicates that aquatic ecotoxicity is the largest relative impact in the US context. The primary driver of the aquatic ecotoxicity is aluminum emissions associated with digestion of wastewater from whey processing and treatment for an effluent containing spilled milk or milk residue removed by CIP. Aluminum in the WWT procedure is used as a coagulant to remove phosphorus via chemical precipitation. The ecotoxicity impact associated with cheese manufacturing represents approximately 0.47 percent of the annual US ecotoxicity impact. It reaches about 0.89% combining cheddar and mozzarella. Because nearly identical result was observed with mozzarella, normalization result on mozzarella is not included here. Terrestrial ecotoxicity and aquatic eutrophication are the next important categories to focus improvement activities. From a manufacturing perspective, these can be mitigated through energy conservation and water conservation/treatment activities.

3.4. Uncertainty analysis

The uncertainty in the results was evaluated using Monte Carlo simulation, available in SimaPro 7.3 software. The ecoinvent pedigree matrix approach was applied to each exchange generated from primary data and then the resultant inventory uncertainty is calculated. These results which are based on a dry solid basis can be used to approximate the impacts on as-sold moisture basis (63.2% solids for cheddar, 51.4% solids for mozzarella). GHG emissions are of notable interest, and on dry-weight basis, the GHG emissions of cheddar and mozzarella are approximately 1.29 and 1.81 tons CO_2e per ton of cheese solids delivered, respectively. The 95% confidence band ranges 0.92 to 1.77 tons CO_2e per ton of cheddar cheese solids delivered, and 1.24 to 2.62 tons of CO_2e per ton of mozzarella cheese solids delivered. For an average moisture content of 36.8% for cheddar as sold at retail, the GHG emission is 0.82 ton CO_2e per ton cheddar delivered with a confidence band of 0.58 to 1.12 tons CO_2e per ton cheddar delivered. Based on an average moisture content of 48.6% for mozzarella as sold at retail, 0.93 ton of CO_2e per ton mozzarella is emitted, with a 95% confidence band of 0.64 to 1.35 tons CO_2e per ton mozzarella delivered. On a dry-weight basis, the freshwater depletion – defined as water removed during the production, but not returned to the same watershed – is 15.7 m^3 per ton of cheddar delivered with 95% confidence band of 12.9 to 19.4 m^3 of water consumed per ton of cheddar delivered. This is equivalent to approximately 9.9 L of water per kilogram of cheddar cheese delivered in the United States. For mozzarella cheese, 13.6 L of water are consumed per kilogram of mozzarella delivered. It should be noted that, even though the same allocation scheme was used for both cheeses, the variability of the co-products for typical cheddar plants and mozzarella plants make direct comparisons of environmental impact results between the two cheese products very difficult and inadvisable.

3.5. Estimated industry scale contribution to US GHG inventory

An estimate of the entire cheese sector impact from farm-gate-to-customer-gate was made through inclusion of the whey and other co-products produced. In reported surveys, mozzarella manufacturing facilities did not produce significant quantities of other cheeses. Cheddar facilities did report some production of Monterey Jack, Swiss, and other natural cheeses. Taking the assumption that all cheeses other than mozzarella had a similar production impact as cheddar manufacturing facilities coupled with whole-plant data, the cheese sector impact was estimated by scaling the study results to national scale. In 2009, about 4.6 million tons of cheese products were produced in the United States (IDFA 2010). The total GHG emissions as sold at retail, including whey and other co-products, are approximately 3.5 million tons CO_2e; cumulative energy demand of 4.7e10 MJ (roughly equivalent to 8.2 million barrels of oil); water consumption of 32 million cubic meters of water.

4. Discussion

From a manufacturing perspective, the study suggests some opportunities to reduce individual plant impacts. First, a focus on plant electricity consumption is explicit since it is the single greatest impact driver. Implementation of energy efficiency best practices should be considered for the refrigeration system, compressed air system, motors, and lighting. Similarly, plant fuel reductions could be realized through improved steam system efficiency and operating practices. Minimizing the amount of water and energy used for CIP can be achieved through using reuse or recovery distribution systems (Baskaran et al. 2003). Finally, careful study of plant specific optimization of the transport distances and the future selection of transport refrigeration systems using low-GWP refrigerants could lead to reduced emissions for the cheese industry.

5. Conclusion

Average impacts in various unit processes (processing, packaging, and distribution) were reported and discussed. For an average moisture content of 36.8% for cheddar as sold at retail, the GHG emission is 0.82 ton CO_2e per ton cheddar delivered with a 95% confidence band of 0.58 to 1.12 tons CO_2e per ton cheddar delivered. Based on an average moisture content of 48.6% for mozzarella as sold at retail, 0.93 ton of CO_2e per ton mozzarella is emitted, with a 95% confidence band of 0.64 to 1.35 tons CO_2e per ton mozzarella delivered.

For the post-farm supply chain, climate change and cumulative energy demand impacts are closely linked with fossil fuel consumption primarily associated with coal mining and combustion. The impacts associated with the electricity supply chain and combustion of other fossil fuels was found to be the most intensive contributor within nearly all impact categories including climate change, cumulative energy demand, ecosystems, human toxicity, and ecotoxicity. Thus conservation efforts to reduce electricity and fuel use will have broad beneficial consequences for environmental sustainability. Truck fleet tailpipe emission is also observed to be a significant impact driver. Thus, seeking the most appropriate transport modes and methods to mitigate the impacts on the environment will deliver financial benefits as well as boost a company's reputation in the minds of consumers.

The regionalized normalization analysis based on an average US citizen's annual cheese production showed that aquatic ecotoxicity stands for the largest relative impact due to aluminum emissions associated with digestion of wastewater from whey processing. Terrestrial ecotoxicity and aquatic eutrophication indicate the next significant impact categories. Therefore, incorporating best practices around heavy metals, phosphorous and nitrogen emissions management can yield good improvements.

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Climate impact of producing more grain legumes in Europe

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ABSTRACT

The grain legumes pea and faba beans are among the relevant alternatives to imported soybeans for livestock feed for growing in the European agricultural systems, but what is the climate impact of an increased European production of grain legumes such as pea and faba bean? In order to estimate the overall climate impact of producing more grain legumes in Europe, we applied Life Cycle Assessment (LCA). The results showed that carbon footprints per kg protein of pea and faba bean in Europe did not vary much for different regions in Europe. Based on FAO statistics and an expert survey (Reckling et al., 2014), it was assumed that an increased European production of grain legumes will decrease the import of soybean cake and decrease the export of wheat from Europe. Taken that into account, results showed a small climate benefit of producing more grain legumes in Europe compared to importing soybeans to Europe.

Keywords: grain legumes, LCA, carbon footprint, soybeans, pea, faba bean

1. Introduction

Soybeans are number one on the world import list of agricultural products based on value (FAOSTAT, 2013); indicating that it is one of the most important agricultural products traded globally. Europe has a net import of 22 mill t cake of soybean and 15 mill t soybeans, but it is not the only region with a high demand for soybean protein. The demand for soybeans for the globally increasing livestock production has seen a dramatic increase during the last decade with China being by far the largest importer of soybeans globally (FAOSTAT, 2013). China has seen a three-fold increase in the import of soybeans during the last decade, with Brazil being the major supplier. Brazil's own consumption of soybeans for the increasing livestock production has also increased and due to the increasing demand, the producer prices of soybeans in Brazil have seen a three-fold increase during the last decade. The European livestock production is vulnerable to increasing prices of soy protein and combined with environmental concerns related to the production and import of soybeans, alternative and more sustainable protein sources are potentially attractive for the European livestock sector. The production of livestock feed is a major contributor to global greenhouse gas emissions, making mitigation options to reduce these important.

Pea and faba beans are relevant alternatives to soybeans in the European cropping systems and livestock diets, since they can be grown across Europe in the different agro-climatic zones. The aim of the present study is to assess the impact on GHG emissions of an increased European production of grain legumes.

2. Methods

We applied a life cycle assessment (LCA) approach to assess the overall climate impact of producing more grain legumes in Europe. The analysis focused on the greenhouse gas emissions or potential global warming impact.

2.1. Estimating carbon footprints of single crops

Studied crops

The basis for the analysis was the cultivation of pea and faba beans in five different agro-climatic zones in Europe, including Sweden (SE), Scotland (SC), Germany (DE), Romania (RO) and Italy (IT). All sites were rainfed. The analysis was based on an expert survey (Reckling et al., 2014) for representative crop rotations with and without grain legumes in the different zones.

Functional unit and system boundaries

The functional unit is 1 tonne harvested crop (DM). However, the greenhouse gas emissions per t harvested protein and per hectare are also presented. The main system boundaries in the present cradle to farm gate study include two main processes; 1) production of agricultural inputs and 2) the agricultural production system. The temporal system boundaries are one year of crop production based on current practice in the five different agro-climatic zones

Estimation of emissions

The characterization factors for the global warming potential (GWP) were based on the IPCC 2007 standards for greenhouse gas emissions (IPCC, 2007). Production data from Reckling et al. (2014), such as crop DM yields and amount of fertilizer, were used for the analysis. The same field operations were assumed for all the sites, which included one ploughing, one harrowing, one sowing, 2.5 pesticide applications, fertilizer applications according to the survey and one harvesting. The GHG emissions related to the fertilizer production were based on Williams et al. (2006). The diesel consumption was based on the field operations and the diesel consumption per field operation was estimated as described by Dalgaard et al. (2002). GHG emissions related to diesel and energy consumption was based on data from the Ecoinvent database (Ecoinvent Centre, 2009). The direct and indirect emissions of nitrous oxide (N_2O) were estimated according to the IPCC guidelines 2006 (IPCC, 2006). NH_3 emissions were estimated according to EEA (2013). N leaching was estimated in accordance with Reckling et al. (2014), who estimated N balances and N losses using the ROTOR model (Bachinger and Zander, 2007). Soil carbon changes related to grain legume production were assumed to be insignificant. The grain legume production was modeled as an individual crop in the crop rotation (Knudsen et al. 2013) , but the interactions with the following crops in the crop rotation was taken into account in terms of a reduced fertilizer use in the following crop of 10 kg N (Plantedirektoratet, 2011) . Furthermore, an increased yield in the following crop of 0.3 t cereal DM per hectare was anticipated and included in the analysis, based on information from Reckling et al. (2014).

2.2. Overall analysis

If Europe starts producing more grain legumes such as pea and faba bean for the European livestock production, the import of soybeans and soybean cakes will be reduced. At the same time the crops being replaced in Europe by grain legumes, will have to be produced somewhere else. According to Reckling et al. (2014), the grain legumes will mainly replace wheat in the crop rotations.

The overall analysis assumes that the production of one hectare of either pea or faba bean will replace one hectare of wheat, previously grown on that land and in that crop rotation. Europe has a net import of soybeans and soybean cake of approximately 37 mill tonnes and a net export of approximately 30 mill tonnes of wheat (FAOSTAT, 2013). It is assumed that an increased production of grain legumes in Europe will decrease the import of soybean cake and decrease the export of wheat from Europe.

Europe mainly imports soybean cake and soybeans from Brazil, but also from Argentina, USA and Paraguay according to FAOSTAT (2013). Since Brazil supplies approximately half of the imported soybeans to Europe, the values from Brazil are weighted accordingly (50%) in the average values while the values from the remaining countries are weighted by 17% each of them. Since part of the wheat production in Europe will be replaced by production of peas or faba beans, the wheat needs to be produced elsewhere. According to FAOSTAT (2013), USA, Canada and Russia are the main wheat exporters globally, and they will presumably increase their production accordingly.

To analyze the overall climate impact of those changes, the carbon footprint of the imported soybean and the wheat production outside Europe is also needed. The inputs and yields from soybean and wheat production in the countries concerned are based on FAOSTAT (2013) (average over the last five years) and FertiStat (2007) and the emissions are estimated based of IPCC (2006) and EEA (2013). Protein content of pea, faba bean and soybean are based on Burstin et al (2011). Transport steps for the import and export of the grains are included in the study.

3. Results and discussion

The carbon footprint of the European grain legumes and the imported grain legumes (soybeans) to Europe is presented (Figure 1). The emissions are given in CO_2 equivalents per t protein for comparison across different grain legumes.

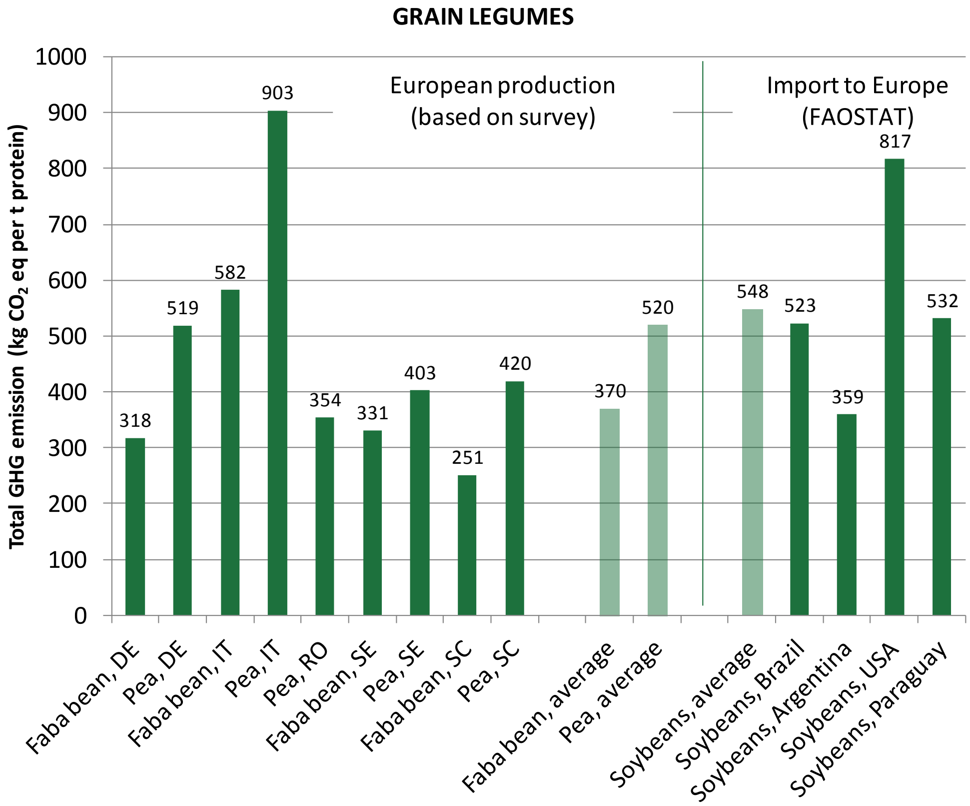

Figure 1. Greenhouse gas emission per t protein for grain legumes produced in five different regions in Europe and soybean imported to Europe. DE, IT, RO, SE and SC refers to the different agro-climatic zones, represented by Germany, Italy, Romania, Sweden and Scotland, respectively.

On average, soybean production outside Europe has a global warming potential of 548 kg CO_2 eq. per t protein. This can be compared to the average for pea and faba bean in our study of 520 and 370 kg CO_2 eq. per t protein, respectively (Figure 1). The higher value for pea is mainly due to lower DM yield and lower protein content in the grains.

The carbon footprint of the wheat produced in Europe (based on Reckling et al., 2014)) can be compared to the carbon footprint of wheat on the world market produced outside Europe (Figure 2). Figure 2 shows that the carbon footprint values are comparable for the wheat production in Europe and outside Europe, despite different yield and input levels. The average values for greenhouse gas emission from wheat production in Europe and outside Europe (Figure 2) are used in the following calculations. Likewise, the values for pea and faba bean production in Europe and for soybean production outside Europe (Figure 2) are used in the overall assessment on the climate impact of producing more grain legumes in Europe.

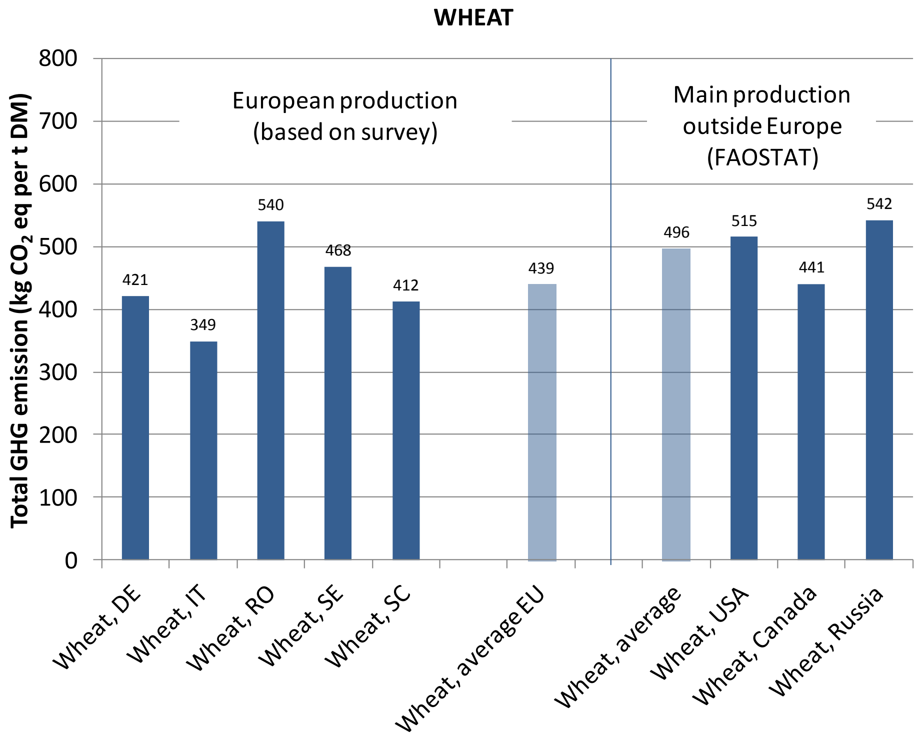

Figure 2. Greenhouse gas emission per t DM for wheat produced in five different locations in Europe and wheat produced by the main wheat exporters outside Europe. DE, IT, RO, SE and SC refers to the different agro-climatic zones, represented by Germany, Italy, Romania, Sweden and Scotland, respectively.

The global flows will be affected when replacing one hectare of wheat in Europe with the production of one hectare of either pea or faba bean in Europe. The wheat formerly produced in Europe will be produced somewhere else. The major global wheat exporters, USA, Canada and Russia are assumed to take over this wheat production. The increased production of grain legumes in Europe, are assumed to replace a corresponding production of protein from soybeans. However, the avoided production soybeans and import of soybean cake to Europe also implies that a certain amount of soybean oil is not produced. The market response to a lower amount of soybean oil on the world market, will, according to Schmidt (2010), be an increase in the production of the marginal vegetable oil, which is palm oil from Malaysia and Indonesia (Schmidt and Weidema, 2008) , and this assumption is used in this work. The division between soybean cake and soybean oil are based on Dalgaard et al. (2008) and the greenhouse gas emissions from palm oil are based on Schmidt (2010).

The overall climate impact of producing more grain legumes in Europe are presented in Table 1 for pea and faba bean.

Table 1. The overall climate impact when replacing one hectare of wheat in Europe with pea or faba bean production, while taking the impact of a reduced soybean import to Europe and an increased wheat production outside Europe into account.

	Kg CO_2 emissions per ha of grain legumes cultivated in Europe	
	PEA	**FABA BEAN**
EMISSIONS		
1 hectare of pea/faba bean production in Europe	296	287
Production of wheat outside Europe	2524	2524
Production of palm oil (to account for less produced soybean oil)	711	1066
TOTAL emissions	3531	3877
AVOIDED EMISSIONS		
1 hectare of wheat production in Europe	-2343	-2343
Soybean production	-329	-493
Soybean, processing to oil and cake[a]	-230	-344
Transport of soybean cake to Europe	-175	-262
Extra carbohydrates in peas/faba beans compared to cake of soybean, avoided cereal production	-733	-610
TOTAL avoided emissions	-3810	-4052
OVERALL climate impact	-279	-175

[a] Dalgaard et al. (2007)

In the assessment it is also taken into account that pea (and faba bean) having the same amount of protein as the cake of soybean that it replaces in the livestock diet, contains more carbohydrates compared to the cake of soybean. This means that it also replaces a certain amount of cereals in the livestock feed – and the production of this cereal can thus be avoided.

The overall impact of producing more grain legumes in Europe showed a small climate benefit compared to importing soybeans to Europe. Approximately 280 kg CO_2 eq. are avoided for each hectare in Europe producing pea instead of wheat. Similarly, 175 CO_2 eq. are avoided for each hectare faba bean produced in Europe instead of wheat.

5. Conclusion

The carbon footprint of pea and faba bean did not vary much over the five different agro-climatic zones in Europe, except from Italy. The carbon footprint of pea cultivated in five different agro-climatic zones in Europe varied from 88-222 kg CO_2 eq. t^{-1} harvested grain DM. On average per kg pea protein, the greenhouse gas emission was 520 kg CO_2 eq. t^{-1} pea protein. For faba bean, the carbon footprint varied from 71 to 165 kg CO_2 eq. t^{-1} harvested grain DM in the different agro-climatic zones in Europe. The greenhouse gas emission per kg protein was on average 370 kg CO_2 eq. t^{-1} faba bean protein.

Based on FAO statistics and Reckling et al. (2014), it was assumed that an increased European production of grain legumes will decrease the import of soybean cake and decrease the export of wheat from Europe. The overall impact of producing more grain legumes in Europe showed a small climate benefit compared to importing soybeans to Europe. Approximately 280 kg CO_2 eq. are avoided for each hectare producing pea instead of wheat in Europe. Similarly, 175 CO_2 eq. are avoided for each hectare faba bean produced instead of wheat in Europe. Thus, the study also illustrates a method for estimating the overall climate effect for a change in production systems.

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Implementing LCA Results for Primary Production in the Agri-Food Sector

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ABSTRACT

Consumer desire for sustainably sourced goods and services is transforming the way our food is grown and how products are made. Quantifying and assessing the environmental impacts of the inputs and outputs of each life cycle phase in primary agriculture production is a complex and resource-demanding task beyond the capabilities and affordability of individual producers. In collaboration with commodity groups and Alberta Agriculture and Rural Development, Quantis Canada completed four environmental footprints. In compliance with the International Organization for Standardization 14040 and 14044 standards, four primary agriculture life cycle assessments (LCA) were completed for Alberta produced canola, potato, egg, and broiler chickens. Hundreds of producers, through online and paper based data collection, participated in the study. This paper explores the use of an LCA approach to systematically identify opportunities to reduce the environmental impacts of primary agriculture production, and to document and communicate environmental performance to internal and external stakeholders.

Keywords: primary agriculture, life cycle assessment (LCA), beneficial management practices (BMP), sustainability, footprint

1. Introduction

Consumer desire for sustainably sourced goods and services is transforming the way our food is grown and how products are being made. In particular, the environmental sustainability is becoming an increasingly important consideration in consumer purchasing decisions. These consumers are those that are active stewards of the environment and are willing to pay a premium for green and socially responsible products (Ross et al. 2010). Environmentally sustainable production processes are becoming more prevalent in the marketplace due to this public pressure as well as shareholder transparency. But more importantly, there is growing evidence that firms that adopt proactive environmental management strategies become more efficient and competitive (Berry and Rondinelli 1998). Calls for corporate social responsibility are coming from investors, insurers, environmental interest groups, financial institutions, and international trading partners. Ultimately, the organizations with in-depth knowledge of how their business operations interact with the environment will remain competitive. In this context, sustainable sourcing is becoming a means of differentiation and access in the agri-food marketplace.

However, quantifying and assessing the environmental impacts of the inputs and outputs of each life cycle phase of primary agriculture production is a complex and resource-demanding task that cannot be afforded by individual producers. The life cycle assessment (LCA) approach systematically identifies opportunities to reduce the environmental impacts of primary agriculture production, and to document and communicate environmental performance to internal and external stakeholders. The objective of this study was to inform producers of the environmental footprint of their production, as well as identify the potential benefits and mitigation strategies producers and commodity associations could implement. The LCA methodology was chosen for this research, as the most comprehensive approach to analyze environmental footprints. To communicate the LCA results, concise and creative communication pieces were developed to inform producers and consumers. In addition, an environmental footprint calculator was developed for Alberta egg producers to measure the footprint of their farm, run different scenarios, and compare results to the benchmark regional average. The information can then be shared with stakeholders, including those who aim to conduct further LCAs of the commodity. This work is also part of an effort to make LCA information accessible and useful for sustainability decision making by producers and the supply chain.

2. Methods

In order to provide the industry with a better understanding of the environmental profiles of current conventional production in Alberta, Alberta Agriculture and Rural Development (ARD), along with Quantis Canada, completed LCAs for Alberta canola, potato, egg, and broiler chicken (including hatching egg) production. In collaboration with the commodity associations, the studies completed environmental assessments of the 2012 practices within the industry using LCA, a framework defined by the International Organization for Standardization (ISO) 14040 and14044 standards (ISO 2006a; ISO 2006b). On farm data was collected from producers through online and paper surveys. Environmental hotspots (those areas with the highest environmental impacts) for different impact categories were identified in each of the LCAs with Impact 2002+ methodology (Jolliet et al. 2003) and associated management recommendations were made. As one of main purposes of the studies was to communicate the results with internal and external stakeholders including non-LCA audiences, a combination of three end-point indicators (human health, ecosystem quality and resources) and two mid-point indicators (climate change and water consumption) were selected.

2.1. Boundaries and Assumptions

The system boundaries chosen for all four commodities identify the life cycle stages, processes, and flows considered in the LCA. The studies assessed the life cycle of Alberta canola, potato, egg and broiler chicken (including hatching egg) production, from the extraction and processing of all farm inputs, to the energy used to the farm gate. The LCA for egg production also included the two Alberta washing and grading facilities that go beyond the farm gate. Within each of these stages, each LCA considered all identifiable upstream inputs to provide a comprehensive view of the production systems. All inputs may therefore be traced back to the original extraction and processing of raw materials used at the various life cycle stages, which include: crop inputs, transportation, field operations, farm utilities, infrastructure, feed production, hatchery, farm operations, egg washing and grading, and waste management.

The study included the diversity of management practices implemented across the industry, and comparisons were made between combinations of practices to identify trade-offs between the various conditions. The project team worked in close collaboration with industry experts to better understand common farm practices and conventional Alberta production of the four commodities. Once the information was reviewed, appropriate scenarios were established for evaluation.

2.2. Data Collection

The quality of the LCA results depends on the quality of the data used as an input in the model. For this study, every effort was made to implement the most credible and representative information available. The life cycle inventory (LCI) was established based upon different data sources.

On farm data was collected to assess the overall environmental impacts of production in Alberta through online and paper surveys. In some cases, information came directly from provincial commodity association records. Data was collected for the 2012 production year. When possible, data was also collected for an additional two years (2011 and 2013) to allow for data gaps or unexplainable data points in the 2012 data. Survey questions covered the entire primary production practices such as acres planted, crop variety, agronomic practice, yield, flock management (diet, feed, and water consumption), manure management, and energy use. Because on farm data can be of variable quality, data from the farm surveys was verified by an industry specialist and compared to industry standards to insure good representativeness. High and low value points for important parameters were tested in sensitivity scenarios but no quantitative uncertainty assessment was realized. Important field emissions as nitrous oxide and ammonia from fertilizer application were based on local Canadian models (Rochette et al. 2008; Sheppard et al. 2010). As the Canadian models of direct nitrous oxide emissions were used in the study, it appeared that uncertainties in the modelling were likely lower in the study than using IPCC Tier 1 specifications.

When no site-specific data were available or the contributions to known impacts were minimal, LCI databases, mainly ecoinvent v2.2, were adapted or used as is. As a last resort, when assumptions were necessary and activity data was not available, stakeholders and experts were consulted to determine specific values.

SimaPro 7.3.3 was used for LCA modeling; it links the reference flows with the LCI database and computes the complete LCI of the systems. The final LCI result was calculated by combining foreground data (intermediate products and elementary flows) and generic datasets, providing cradle-to-gate background elementary flows to create a complete inventory.

3. Results

The four LCAs were assessed based on the environmental scores of Alberta production through five end-point indicators: human health, ecosystem quality, resources, climate change, and freshwater consumption, as well as water stress for potato production. Mid-point indicators of Impact 2002+ were also analyzed. It is important to note that LCA estimates relative, potential impacts rather than direct measurements of real impacts, and that results and conclusions should be considered applicable only within the scope of the study. The ecosystem quality indicator expresses the composite score of eight midpoint categories which measure toxicity effects of pesticides.

3.1. Alberta Canola Production

Crop inputs are responsible for the greatest proportion of impacts for all categories. The efficiency of the farm input use (e.g. fertilizers, pesticides, seeds) and land use is directly proportional to canola production yields. Electricity and gas consumption on farm varied significantly from one producer to the next showing potential room for improvement.

3.2. Alberta Potato Production

Crop inputs are responsible for the greatest share of environmental impacts for all indicators, except water stress, where irrigation is the main contributor. The production and emissions associated with synthetic fertilizers are significant contributors to impacts for climate change, human health, and resources. The proper management of synthetic fertilizers is therefore crucial to achieve considerable environmental impact reductions. The most variable parameters between farms are electricity consumption for potato storage and irrigation requirements. This is partly explained by the fact that not all producers store their potatoes and by the different efficiencies of storage facilities between producers.

3.3. Alberta Egg Production

Feed production was identified as the largest contributor to all environmental impacts, followed by farm operations. Fertilizer production and application, and land occupation are major causes of high environmental impact in feed production, while on farm energy use and manure management are responsible for most of the environmental impacts for the farm operations. Preliminary results of the enriched cage system indicate equivalent feed efficiency of the birds and globally slightly lower environmental impacts.

3.4. Alberta Hatching Egg and Broiler Chicken Production

The feed production stage is the main contributor to all impact categories, except climate change. The main impacts for climate change come from farm operations, specifically the energy used for heating the barn. Coal heated barns, which consume more fuel per functional unit compared to other energy sources, are responsible for causing a higher impact in this category. Hatching egg production is the third most important contributor to the life cycle impact of broiler production.

4. Discussion

LCA methodologies and background databases are becoming more accurate by depicting impacts at a regional level through environmental modeling. At the same time, agricultural technologies are evolving and becoming increasingly precise and efficient. As such, the four Alberta agri-food LCAs establish a benchmark, allowing industry to be prepared for market demands and reflect the new technological changes in the industry. With a credible baseline in hand, representing provincial average production systems, specific attention was given towards management recommendations. The LCA model was used to evaluate the environmental benefit of implementing beneficial management practices (BMP), focusing on parameters that producers have the ability to influence and/or that the LCA identified as a hotspot. A summary of the commodity-specific LCA results and associated BMP recommendations are provided below in Table 1.

Table 1. Environmental footprint results and BMP recommendations by commodity

Commodity	**Life cycle stage with largest footprint (descending order)**	**Primary BMP recommendations**
Canola	Crop inputs Farm utilities and infrastructure Field operations Waste management Transportation	• Adoption of 4R Nutrient Stewardship to manage nutrient sources • Soil carbon sequestration from conservation tillage
Chicken	Farm operations Feed production Hatching egg Hatchery Transportation	• Phasing out coal barn heating for natural gas or biomass heated barns • Encouraging use of the industry's efficient feed conversion ratio
Egg	Feed production Farm operations Hatchery Transportation Washing and grading	• Implementing on farm energy efficiency measures such as well-designed ventilation systems • Soil carbon sequestration from conservation tillage
Potato	Crop inputs Farm utilities and infrastructure Irrigation Field operations Waste management	• Adoption of 4R Nutrient Stewardship to manage nutrient sources • Ongoing improvement of irrigation infrastructure and delivery equipment, such as a variable rate irrigation system

4.1. Alberta Canola Production

The yield per acre improvement constitutes an important lever for producers considering that the efficiency of the farm input use and land use is directly proportional to canola production yields. The synthetic fertilizer production and emissions drive farm input impacts for all categories except ecosystem quality; therefore, improving fertilizer use efficiency is crucial to the reduction of environmental impacts through BMPs such as the 4Rs nutrient stewardship system - using the right fertilizer, right rate, right time and right place.

On-farm energy efficiency measures can be a significant potential mitigation area to reduce the environmental impacts of canola production. Reducing the number of passes and opting for operations that are less fuel intensive could reduce the impacts of field operations. It is important for industry to give priority to the high impact areas (i.e. hotspots indicating significant contribution to the total footprint) to ensure that targeted investments in BMPs yield significant environmental paybacks. Fertilizer management and tillage management BMPs are another potential mitigation strategy for reduction of environmental impacts. BMPs related to better fertilizer management are based on the "4R" nutrient stewardship principles. Potential greenhouse gas (GHG) emissions reductions ranged from 2% to 23% for fertilizer management. Depending on the tillage system, a potential reduction of impacts can be achieved by adopting conservation management practices such as no-till/zero-till and minimum/reduced systems. It can therefore be concluded that Alberta canola producers can manage their environmental impacts and footprint to a certain extent by continuing to adopt BMPs.

4.2. Alberta Potato Production

On-farm energy efficiency measures for potato storage and water pumping are important levers for growers, with significant potential for reducing the environmental impacts of potato production. Reducing the number of passes and opting for new measures, technologies, tools, or machinery that increase fuel efficiency could significantly reduce the impacts of field operations. Continued work on reducing water loss from sprinkler systems with variable rate irrigation and low pressure distribution systems at the farm would lower freshwater consumption. The "4Rs" of fertilizer management—the right product, right rate, right time and right place—could significantly reduce GHG emissions as well.

The potato growers have been adopting a number of BMPs related to fertilizer management, tillage management, irrigation management, and on-farm management. Over the past decade, Alberta potato growers have made steady progress in adopting many of the BMPs identified to reduce the environmental footprint of potato production. BMPs related to better fertilizer management were based on the 4R Nutrient Stewardship system. Potential impact reductions ranged from 1% to 25% for climate change and 11% and 13% for freshwater consumption. It can therefore be concluded that Alberta potato growers have opportunities to reduce the environmental footprint of their production through continuous improvement and adoption of BMPs.

4.3. Alberta Egg Production

Considering that the feed production stage is the main contributor to all impact categories, optimizing feed efficiency is a key lever to reducing the egg footprint. Furthermore, feed is one of the major contributors to the cost of production. Nonetheless, collected surveys indicate that feed efficiency can vary significantly between farms. The next step for industry should be to focus on working with nutritionists and published literature to produce appropriate recommendations to producers. Another significant impact is the production of feed, therefore promoting BMPs back to the feed producers could lower the egg footprint (lower impact of producing the feed will lead to a lower impact of the egg).

Due to the importance of energy consumption in the farm operations stage, energy reduction measures were investigated. Energy efficiency measures at farm could save up to 21% on electricity consumption. Implementation of renewable energy in the electricity grid mix or at farm could be also be beneficial, but to a lesser extent. The difference in the results between the baseline scenario and the use of 30% of renewable energy in the grid mix did not exceed 3%. Feed transport is the major contributor to transportation impacts. Therefore, increased proportion of farm-grown and locally grown feeds should be used for feed formulation to avoid the transportation of feed crops from the United States and other provinces.

4.4. Alberta Hatching Egg and Broiler Chicken Production

Optimizing feed efficiency is a key factor to reducing the environmental footprint of broiler production. Feed is one of the major contributors to the cost of production. Similar to egg production, collected surveys for this project indicate that feed efficiency can vary significantly between farms. As suggested previously with regard to eggs, a next step could be to focus on working with nutritionists and published literature to produce appropriate recommendations for producers. Hence, broiler producers would benefit from being informed regarding the research on feed efficiency in order to align all factors to favor a better feed conversion ratio (FCR).

Promoting beneficial practices to feed producers could significantly lower the broiler footprint (lower impact of producing the feed will lead to a lower impact for the broiler). In particular, an increased understanding and the adoption of BMPs for fertilizer use in feed production could be further investigated. Also, prioritizing on farm energy efficiency measures for space heating, phasing out of coal furnaces to natural gas powered furnaces, or implementing biomass or litter capable furnaces, would be beneficial to lower GHG emissions.

Being very similar to its broiler counterpart, hatching egg production can orient its priorities accordingly, particularly by reducing the FCR and using feed with a lesser impact. One important aspect to work on is hatchability, as only 79% of fertilized eggs are sold as broiler chicks. Reducing the loss rate would greatly improve the impact of this stage.

The key impact findings of this study can be addressed by adoption of BMPs related to feed, energy, and operational changes. GHG emission reductions can result in 1% to 5% for feed, and 1% to 33% for energy related BMPs. Hence, broiler chicken producers have considerable leverage over their environmental impacts by implementing certain BMPs.

5. Conclusion

Alberta Agriculture and Rural Development (ARD) commissioned the LCA of four commodities in order to better understand and quantify the environmental impacts of the province's agri-food industry. ARD is dedicated to ensuring that Alberta's agricultural producers remain competitive, adaptive, and responsive in the marketplace. With the studies completed, highlights and key findings have been documented for easy-to-communicate two-pager factsheets that will be distributed to industry associations and producers in an effort to raise awareness and highlight opportunities to reduce environmental impacts, especially GHG emissions and freshwater consumption. In addition, a calculator for producers to assess the footprint of their farm is available to all Alberta Egg Farmers online. This will enable the egg producers of Alberta to benchmark their farms and identify farm specific reduction opportunities. Later, stewardship initiatives to help producers implement environmentally, technically, and economically sound BMPs on their farms may be launched.

This holistic and systematic environmental assessment helps to identify environmental hotspots of primary production in Alberta and propose potential mitigation strategies that could improve environmental performance of the systems. Indeed, an opportunity to reduce GHGs and the shift by certain European jurisdictions toward a multi-criteria environmental food product labeling system has led to the creation of sustainability-based markets.

This study provides industry with a better understanding of the environmental profiles of conventional production in Alberta. The study also aimed to set out a scientifically robust and transparent environmental assessment of current practices of the industry. This Alberta benchmark will provide a way to measure improvements and focus on identifying opportunities to enhance the environmental performance of Alberta's agri-food industry. In addition to providing a benchmark, the work established a protocol for data collection and an approach for continuous improvement throughout the industry. The initiative will also serve as a model for the analysis of other local and international agricultural commodities.

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Environmental Life Cycle Impacts of High Oleic Soybean Oil Used for Frying

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ABSTRACT

Life cycle assessment was used to quantify the environmental impacts and identify supply chain hotspots for high oleic soybean oil (HOSO), conventional soybean oil (CCO), hydrogenated soybean oil (HSO), high oleic canola oil (HOCO), and conventional canola oil (CCO) for use in US restaurant fryers. On a cradle-to-grave basis, environmental impacts for fryer oil are a function of oil use rate, manufacturing burdens, and restaurant management behavior. Results identify 30%-45% reductions of impacts for high oleic oils relative to their conventional counterparts for the relevant impact categories evaluated. Impacts are similar for HOSO and HSO based on similar use-phase performance. The LCA highlights the importance of oil performance in use, nitrogen fixation of soybeans, allocation assumptions, and waste disposal considerations of the oil after use. Using LCA demonstrates that HOSO not only offers nutritional and functional benefits but allows restaurants to improve the environmental footprint of their oil ingredients.

Keywords: Soybean Oil, High Oleic Soybean Oil, Canola Oil, High Oleic Canola Oil

1. Introduction

Many markets are experiencing a trend towards sustainability. The fryer market is no exception as consumers demand healthier options, causing restaurants to respond in kind, but without wanting to sacrifice cost, taste, or quality. For more than 50 years hydrogenated oils have been the main stay in the fryer oil market, providing increased stability, long fry life, and low cost relative to conventional oils (AHA, 2010). However, health concerns about trans-fats have caused regional bans on their use including bans in New York, Philadelphia, and California (Tavarnise, 2013). Top chefs and restaurant chains are looking for alternative oils that are just as stable and cost effective but without trans-fat and lower saturated fat content. Recently, the FDA announced a preliminary decision to remove partially hydrogenated oils from the 'Generally Recognized As Safe' (GRAS) list which will continue to boost the need for alternative highly stable oils. To stay in the market, oil suppliers are met with the task of supplying drop-in replacement products which can meet the nutritional demands without increasing costs. A switch to conventional oils may meet nutritional requirements, but they do so at the expense of fry-life. High oleic soybean oil is now available in the U.S. market which provides both the desired functionally (fry-life & taste) in addition to eliminating trans-fats and reducing saturated fat content. To complement the nutritional and performance benefits of high oleic frying oils, this study takes a holistic approach, using life cycle analysis, to quantify the environmental impacts of high oleic frying oils in restaurant fryer applications relative to those for conventional and hydrogenated oils. The goal of the study is three fold:

- Quantify the importance of functionality and oil performance in the use-phase
- Identify hotspots within the fryer oil supply chain;
- Understand the importance and sensitivity of the assumptions used in the LCA

2. Methods

Attributional life cycle methodology is used to evaluate the environmental impacts associated with frying oil use in restaurants for conventional soybean oil (CSO), high oleic soybean oil (HOSO), hydrogenated soybean oil (HSO), conventional canola oil (CCO), and high oleic canola oil (HOCO). The study focuses on U.S. restaurant fryer oil applications. Oil availability, consumer demands, restaurant practices, oil transportation, and, of course, seed agronomics may all vary with geography. No comparative assertions are intended. Comparisons among different seeds are made to highlight hotspots within the supply chains for the US market.

2.1. Functional Unit

The functional unit for this system is based on days of fryer oil use in a restaurant. As restaurants have many variations in terms of food throughput, the types of foods fried, and hours of operation, more than one scenario is required. A base case, Scenario 1, assumes two days of fryer use and a 4% loss rate of oil to the food per day. An alternate case, Scenario 2, assumes six days of fryer use and a 10% loss rate of oil to the food. Both cases follow fry-life test results where high oleic oils and hydrogenated oils have increased stability, allowing twice the fry-life of conventional oils based on polar compound generation rates (see Section 2.4.4). For each scenario, high oleic and hydrogenated oils are charged once, while conventional oils require two charges. Top-off is assumed to occur each day if the oil has not been changed. All cases assume four (4) 22.7 kg (50 lb) fryers are used at the restaurant. These two scenarios are not all-inclusive, but provide perception into the impacts of food absorption rates and fry-life. Extended fry-life increases the number of times oil must be topped-off while higher absorption rates increase the amount of top-off oil required. Since high-oleic and hydrogenated oils require half as many charges, one less day's top off quantity is required in each scenario for these oils relative to conventional oil. Table 1 summarizes the oil use rates and wash cycles for each type of oil for each scenario.

Table 1. Fryer Oil Use Scenarios; Scenario 1: Two Days Fryer Use, 4%/day Oil Loss Rate; Scenario 2: Six Days Fryer Use, 10%/day Oil Loss Rate

		Scenario 1		**Scenario 2**	
		Conventional	**High Oleic / Hydrogenated**	**Conventional**	**High Oleic / Hydrogenated**
Oil, Charged	kg	181.4	90.7	181.4	90.7
Oil, Top-off	kg	0	3.6	36.3	45.4
Oil, Total	kg	181.4	94.3	217.7	136.1
Oil, Spent	kg	174.2	87.1	163.3	81.6
Wash Cycles	-	2	1	2	1

2.2. System Boundaries

The system boundaries for this study are illustrated in Figure 1. They include agricultural inputs, farming operations, seed processing, soybean and canola oil refining (and hydrogenation when required), transport along the supply chain as well as transport for the refined oil to a warehouse local to the restaurant, oil use in the restaurant, washing of the fryers, and spent oil disposal. Key exclusions from the system boundary are as follows:

- Health impacts (Scope of the study is on environmental impacts)
- Electricity to operate the fryers at the restaurant (equivalent in all cases)
- Conversion of spent oil at EOL to alternative products such as biodiesel, animal feed, or boiler fuel (This study uses a restaurant perspective – see section 5)
- Transportation from the warehouse to the restaurant (deemed negligible)
- Oil packaging (would increase with increased total oil use – i.e. higher for conventional oils)
- Indirect effects such as indirect land use change (attributional modeling employed; would increase with increased total oil use – i.e. higher for conventional oils)
- Infrastructure
- Sequestered carbon in oil (Study assumes all biogenic carbon sequestered in the oil is released via digestion (absorbed oil), combustion (spent oil), or biodegradation (spent oil).)

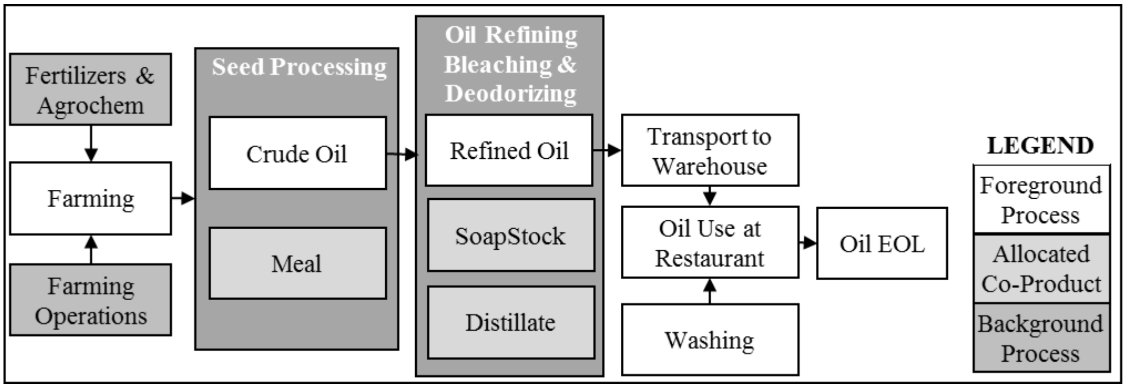

Figure 1. System Boundaries for Soybean Oil or Canola Oil.

2.3. Impact Categories

Based on the agricultural processes involved, the following impact categories were identified as relevant and included in this study: Climate Change Potential (CCP), Eutrophication Potential (EP), Acidification Potential (AP), Land Use, and Non-Renewable Energy Demand (NRE). The TRACI 2.0 impact assessment method is used for CCP, EP, and AP based on the U.S. focus for the study. NRE is calculated using the Cumulative Energy Demand (CED) Impact method. The CED model available in SimaPro™ software has been corrected to quantify all energy flows on a higher heating value (HHV) basis. Land Use is included and modeled using the impact category available in the ReCiPe Midpoint (Heuristic) model. Water use is both relevant and of interest, but is outside the scope of this study as this would require more regional information to provide meaningful results.

2.4. Process Models

2.4.1. Farming: U.S. Soybeans, Canadian Canola

U.S. soybean supply is virtually all domestically grown. A detailed life cycle analysis from the United Soybean Board (USB) is used as the basis for the soybean farming life cycle model (Omni-Tech, 2010). From 2007-2012, more than 65% of the canola oil supply to the US is imported, with most of that from Canada. A detailed life cycle assessment for the production of canola from the Canola Council of Canada (S&T2, 2010) is used which is deemed representative for the US supply chain and for US canola farming (S&T2, 2010; USDA, 2014). Ammonia air emissions and phosphorus and nitrate water emissions were adjusted or added such that both datasets used the same methodology for calculation (Nemecek & Kagi, 2007). Dinitrogen monoxide, a key field emissions contributing to CCP, has been updated since the original USB report from 0.35g N_2O per kg soybean based on DAYCENT modelling to 0.682 g N_2O per kg soybean based on actual field measurements in Minnesota. No carbon sequestration is included in either seed model as all carbon is assumed to be converted back to carbon dioxide at the end-of-life through digestion, combustion, or degradation.

2.4.2. Seed & Oil Processing

Data from the USB is used to model the burdens associated with soybean processing (Omni-Tech, 2010). Specific canola processing data is not available in the Canola Council of Canada LCA. Detailed and consistent data for a European rapeseed mill is available in the literature (Jungbluth, N., et. al., 2007; Schmidt, 2007). For this study, canola mill yields, solvent use, and energy consumption are taken from the thesis by Schmidt.

Since mill processing results in both oil and meal co-products allocation is required. This study uses economic allocation of the oil and meal for both soybeans and canola. For soybeans, average price and production data of US meal and oil from 2007-2011 per the USDA NASS oil crops yearbook (USDA, 2014) is used. For canola, price and production data are taken from Statistics Canada(StatCan, 2014). Sensitivity analysis for allocation methods is included in this study.

For high oleic oil allocation purposes, pricing for crude high oleic soybean and canola oil is assumed to be \$0.132/kg (i.e. \$0.06/lb) higher than conventional oil. No price differences are included for meal.

Refining, bleaching and deodorizing are required for both soybean oil and canola oil food applications. Both are modeled using the same oil refining process data (Schmidt, 2007). In addition to refined oil (98.7% by mass), soap stock and a distillate co-product are produced. Spot pricing for the co-products is used for economic allocation. The refined oil receives roughly 97% of the allocation.

Hydrogenation was modeled by accounting for energy (100 kg steam, 13 kWh electricity), Hydrogen (72 m3 at STP), refined soybean oil (1010 kg), and nickel catalyst (0.3 kg) required per tonne of hydrogenated soybean oil (Chakrabarty, 2009). Allocation for HSO is not affected by its higher price since hydrogenation occurs after both the milling and refining steps.

2.4.3. Transport: Refined Oil to Warehouse

The transport for refined oil to the restaurant is simplified to capture just the main portion of travel from the refinery to the warehouse local to the restaurant. As a base case, the restaurant is assumed to be in New York City, NY, requiring 900 miles of transport for soybean oil (i.e. distance from Rock Island, IL) and 2200 miles for canola Oil (from Prince Albert, Saskatchewan). Local transport from warehouse to the restaurant and backhaul is ignored. Sensitivity analysis on transport assumptions is included.

2.4.4. Use-Phase: Fry-Life and Washing

DuPont internal fry-life tests were performed for high oleic soybean oil relative to conventional soybean oil. For HOSO, oil stability, as measured by the composition of polar compounds in the oil reaching 25% during prolonged time at frying temperature, was at least twice that of conventional soybean oil. Actual fry-life is a function of restaurant cooking demand and restaurant behavior. This study assumes that the restaurant cooking demand results in either 2 or 6 days of fry life for HOSO and, half of that, 1 or 3 days, for conventional soybean oil. HOCO and CCO fry-life performance is estimated to be consistent with HOSO, and CSO, respectively. HSO is estimated to have the same fry-life as the high oleic oils (Pambau, et.al, 2010).

The washing procedure used in this study is based on Stratas Foods Fryer Tips (Stratus, 2009). Two fryer volumes of water are used for washing including 1 cup of vinegar per five gallons of water. The energy associated with the washing step is based on heating the water from 50°F to 200°F with electric heat. The same washing procedure is assumed for all oils. Washing is assumed to be done every time the oil is changed.

2.4.5. Spent Oil

Spent oil has several potential end-of-life scenarios after use in the restaurant. For instance, the spent oil may be used for animal feed, may be processed and used as fuel in a boiler, or may be processed to yellow grease and used as a feedstock to biodiesel generation. However, the restaurant typically does not control the end use for the oil. As it leaves the restaurant, the spent oil has free fatty acids generated during use and contains water and animal fats as well as other impurities. In the past, restaurants would need to pay for spent oil waste removal. Currently, many restaurants are having the oil collected without charge or credit, while some may receive a small credit. From the restaurant's perspective, an appropriate method to model the spent oil is via economic substitution. For the base case, we assume no burdens and no credits for the spent oil, so no adjustments, additions, or credits to the model are required to address the end-of-life (EOL) for the spent oil. Sensitivity analysis is included for modeling of spent oil.

Of note, no carbon sequestration is assumed in the production of soybeans or canola, so no carbon emissions associated with the spent oil need to be addressed at end-of-life. All carbon in the oil is assumed to return to the atmosphere as carbon dioxide via digestion, combustion, or decomposition.

2.5. Auxiliary Models

Throughout the process models, several background LCA models are required for electricity and fuel supply, minor chemical, fertilizers, farming operations, etc. In general, US-EI 2.0 database models are used. These models are based mainly from ecoinvent 2.2, where all electricity flows have been replaced with a US average electricity mix. Foreground models use region specific electricity.

3. Results

For each impact assessment, the burdens associated with different steps of the supply chain were segregated to identify their contribution to the total burden. The term ‘Farming’ includes the production, transportation, and application of fertilizers, soil emissions, and the on-farm energy. ‘Milling/ Refining’ includes all processing energy and impacts allocated to the oil including milling, solvent extraction, degumming, neutralization, bleaching, and deodorizing. For HSO, impacts associated with the hydrogenation are included as well. ‘Transportation’ refers only to burdens of transporting the refined oil to a warehouse local to the restaurant. ‘Washing’ refers to cleaning the fryers at the restaurant each time the oil is changed. For each oil type, Figures 2a-2e show the breakdown of impacts by these process areas.

The results presented are based on several assumptions and methodologies. Sensitivity analyses are required to understand the importance of these assumptions and to test the robustness of the magnitude of the impacts and the trends identified in the results. These are presented for fry-life and oil absorption, oil transportation from the warehouse, allocation methods, and spent oil allocation.

3.1. Scenario 1; 2 Days Fryer Use, 4% Oil Loss Rate

In general, under the assumptions in this study, environmental impacts follow the functionality of the oil. The increased oil stability due to the high oleic content or hydrogenation uses less oil, results in lower transport requirements and requires fewer washes. HOSO and HOCO achieve these savings at the expense of slightly higher allocation at the mill stage due to a higher valued oil product. HSO requires additional processing following the refining steps. Figures 2a-2e show the impacts broken down by process step for CCP, NRE, AP, EP, and Land Use, respectively. Across all impact categories, the increased oil stability provided by the high oleic oils and HSO result in roughly a 42%-47% reduction in impact relative to conventional oils.

3.1.1. Climate Change Potential

As shown for Scenario 1 in Figure 2a, CCP is dominated by the farming step for each oil type. For soybean oils in scenario 1 (2 days frying, 4% oil loss rate), farming contributes 53%-57% of the total impacts. Field N_2O emissions account for half of the soybean farming CCP burden, while emissions during quicklime production account for 24%, and emissions from fuel consumption at the farm account for 16%. Milling adds another 28%-31% to the overall CCP. HSO has slightly lower climate change impacts than HOSO. For CCP, the increased impacts from economic allocation for the higher priced HOSO are marginally higher than the burdens from hydrogenation for HSO. Although the overall CCP impacts for canola oil are higher than those for soybean oil, two-thirds of the difference is due to oil transport to the warehouse. Canola oil has higher CCP from farming due mainly to increased fertilizer use (particularly N-based fertilizers). Canola oil has 40% lower impacts in the milling process per kg oil relative to soybean oils.

3.1.2. Non-Renewable Energy Demand

Figure 2b shows the NRE impacts for Scenario 1. For soybean oils, milling and refining impacts account for 44%-50% of the total NRE impacts, while farming only accounts for 26-29%. Washing impacts are more important for NRE than for CCP, accounting for 18%-21% of the total NRE. Roughly 50% of the soybean farming NRE comes from on farm energy requirements. The impacts from HSO are 3% higher than those for HOSO. For canola, fuel use at the farm accounts for 45%, while the production of N-based fertilizers accounts for 44% of canola farming NRE impacts. Differences between soybean and canola oils are more pronounced for NRE as compared to CCP, mainly due to the higher fertilizer inputs for canola.

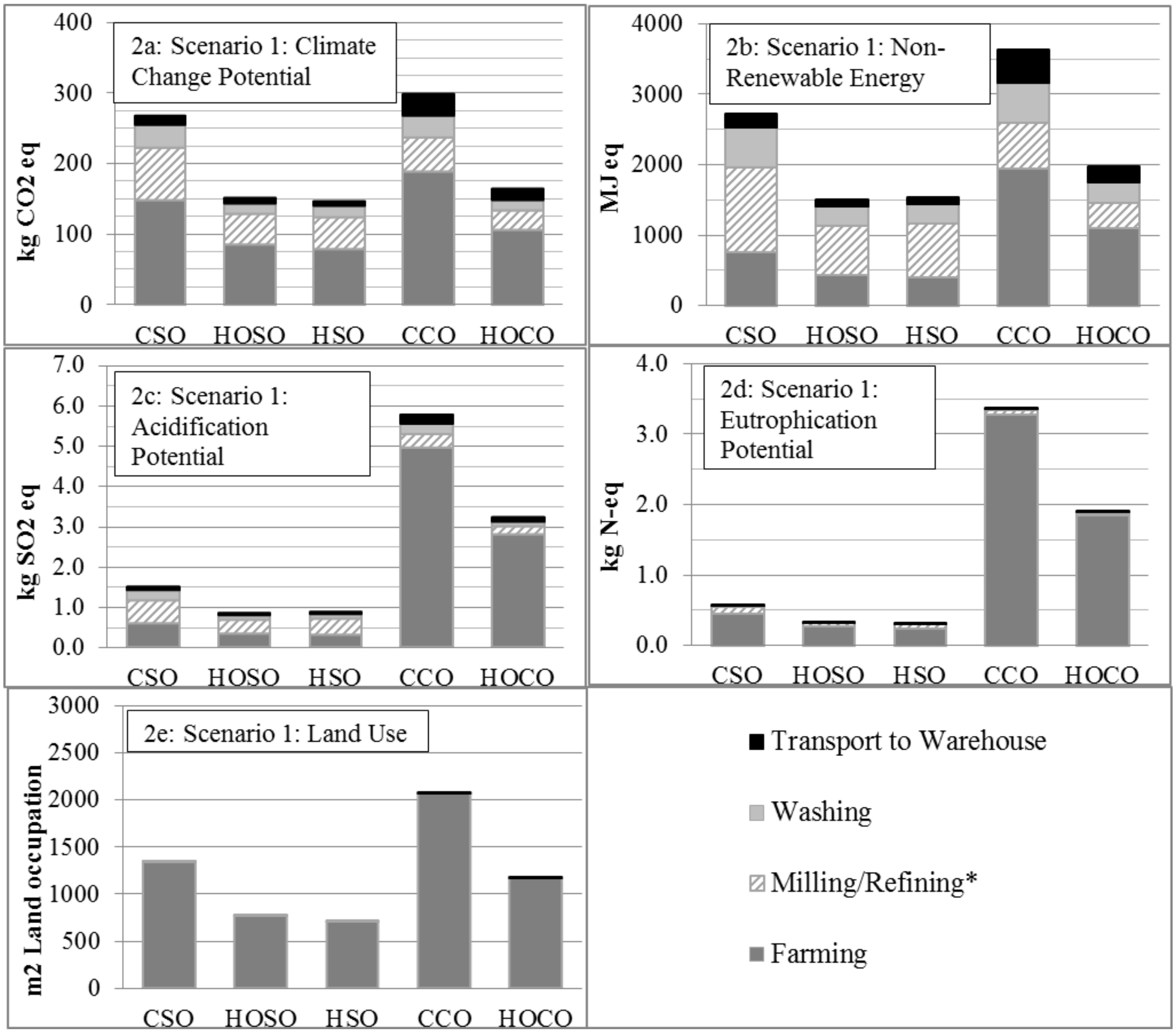

Figure 2a-2e: Impacts by Process Step and Oil type for Scenario 1. Impacts evaluated include Climate Change Potential, Non-Renewable Energy Use, Acidification Potential, and Eutrophication Potential, and Land Use.

3.1.3. Acidification Potential

Ammonia, nitrogen oxide, and sulfur dioxide are the three main emissions affecting acidification. In the frying use supply chain, these are released as field emissions, from fuel combustion throughout the supply chain, and from production of some N-based fertilizers. For soybean oils, where N-fertilizer demand is low because of the N-fixation properties of the legume, the fuel combustion pathway dominates. As shown in Figure 2c for Scenario 1, farming impacts are 38% of the total AP impact for soybean oils, with less than 12% from direct field emissions. For canola oils, field emissions of ammonia drive the AP impacts since fertilizer use is much higher, leading to much higher AP impacts than soybean oils. More than 85% of the AP impacts for canola oils come from farming with over 66% from direct field emissions.

3.1.4. Eutrophication Potential

Eutrophication is driven mainly by releases of nitrates, ammonia, phosphates, and phosphorus in this supply chain. Both soybean and canola farming play the major role for their respective oil co-products. As shown in Figure 2d, for soybean oil, farming accounts for 78%-80% of the total impacts, with nearly 30% from field emissions. The impacts from canola fertilizer requirements are even more pronounced for EP than seen above for AP, as farming accounts for more than 97% of the total EP potential with 72% from direct field emissions.

3.1.5. Land Use

Not surprisingly, agricultural land use is governed by farm yields, oil content, and the amount of soybeans or canola allocated to the respective oils. For CSO, at a yield of 2768 kg soybeans / ha, an oil content of 19.5%, and an economic allocation factor of 39.42%, 1368 kg crude soybean oil are produced per hectare. For CCO, at a yield of 1550 kg canola/ha, 42.6% oil content, and a 75.01% economic allocation factor, 880 kg crude canola oil are produced per hectare. Due to use of economic allocation for the base case, where high oleic oils are priced $0.132/kg higher than conventional oils, a higher allocation rate of seed is applied to HOSO and HOCO per kg of oil used. Therefore, for Scenario 1, although oil use rate drops by 48% due to high oleic oil stability, land use decreases only 43%. Based on economic allocation, canola oils require over 50% more land per 2-day fryer use in Scenario 1. Figure 2e shows the land occupation for each oil type for Scenario 1.

3.2. Sensitivity Analysis: Scenario 2; 6 Days Fryer Use, 10% Oil loss rate

In general, Scenario 2 impacts are higher than those from scenario 1, but with less differentiation between conventional and high oleic oils due to higher oil adsorption rates to the food. The benefits of additional oil stability are not realized for the oil that is absorbed. The importance of washing is diminished for Scenario 2 since washing burdens don't change compared to Scenario 1, but oil use rate increases. For a given oil type, the other relative impacts of farming, milling, refining, and transport are similar to Scenario 1. Impacts for conventional oils increase roughly 17-19% compared to Scenario 1while high oleic impacts increase 40%-42%. HSO impacts increase 37%, slightly less than high oleic oils due to economic allocation.

3.3. Sensitivity Analysis: Transportation Assumptions

Transportation of the oil is clearly a function of location. Selecting cities closer to the refinery would reduce the magnitude of the oil transport to the warehouse. Selecting different cities may also change the relative impacts of canola based oils versus soybean based oils, but would not affect the relative impacts of high oleic and conventional oils. In the base case, transport had the highest impacts on CCP and NRE, but still only account for 5-13% of the total impacts. Impacts for transportation for three other U.S. cities are presented to show the change in magnitude of impact due to distance and the relative impact across seed types. Table 3 shows the CCP impacts for both transportation and for the functional system for Scenario 1 for each city. The relative impacts from transport range from 1%-10% for soybean oils and 5% to 13% for canola oils. For Seattle, where the transport distance is greater for soybean oil than canola, the overall CCP burdens from Scenario 1 for the different seed types become equivalent instead of favoring soybean oil. For NRE, Scenario 1 transport impacts range from 1%-14% for soybean oils and 7%-15% for canola Oils depending on restaurant location. Unlike for CCP, the Seattle location still favors soybean oils over canola oils, but the relative difference is decreased from 30% to 15%. Transport has much less impact for AP, EP, and Land Use and does not result in any significant changes in relative impacts among oil types for any restaurant location evaluated.

Table 3. Transportation Sensitivity: Transportation and Total Impacts for Scenario 1, CCP, at various U.S. Cities

Restaurant Location	Distance, km			CCP, kg CO2 eq.				
	Soybean	Canola		CSO	HOSO	HSO	CCO	HOCO
New York City, NY	1450	3540	Transport	13.1	6.8	6.8	32.1	16.7
			Total	**268**	**151**	**146**	**300**	**165**
Seattle, WA	3080	1820	Transport	27.9	14.5	14.5	16.5	8.6
			Total	**283**	**159**	**154**	**285**	**157**
Los Angeles, CA	2990	4410	Transport	27.1	14.1	14.1	40.0	20.8
			Total	**282**	**158**	**154**	**308**	**169**
Chicago, IL	270	2210	Transport	2.4	1.3	1.3	20.0	10.4
			Total	**257**	**146**	**141**	**288**	**159**

3.4. Allocation Methods

Central to LCA methodology is how to address the burdens associated with co-products. Allocation choices affect both the magnitude of impacts for all oil types and the relative impacts between soybean oils and canola oils since soybeans and canola have significantly different oil and meal compositions. The referenced USB LCA study for soybeans used mass allocation of soybean oil and soybean meal at the mill. However, the two co-products have significantly different value. Further, soybean meal is actually valued on its meal protein content, not just its overall mass since it may be sold with and without the hulls. This study has used economic allocation as the base case to address the difference in value of the two co-products. This prevents favoring soybean oils relative to canola oil based on the higher meal content in soybeans. A downside to economic allocation is price fluctuation. Since oil and meal serve different markets, they do not necessarily vary in the same manner. However, these fluctuations are dampened by averaging across multiple years of production.

For comparison to the base case economic allocation method (ECON), two additional allocation techniques are investigated, including mass allocation (MASS) as used in the USB report, and mass allocation between the meal protein content and the oil content in the seed (MASS-MP). The MASS-MP basis combines the simplicity of mass allocation with the valued aspects of the two co-products. Process subdivision was not credible for the mill stages due to both processing and energy integration.

Meal protein concentrations for soybeans from the United Soybean board and have been averaged across the 2007-2010 time horizon at 44.2% of soybean meal (USB, 2010). Meal protein content for canola meal is reported at 36% (S&T2, 2010). Results for CCP based on scenario 1are shown for each oil type and each allocation method in Figure 3 along with the mill allocation factors for each allocation method. Mass allocation results are 32%-45% lower than economic allocation for CCP. Reductions are higher for soybeans where more meal is produced relative to oil and for high oleic oils where the oil has higher value. While still lower, the MASS-MP results are much closer to economic allocation results since meal is priced on protein content. For each impact category evaluated, using mass allocation or MASS-MP allocation generally reduces the overall impact for fryer oil use, increases the differentiation between soybean oil and canola oil, and results in HSO being consistently higher than HOSO due to the impacts from hydrogenation

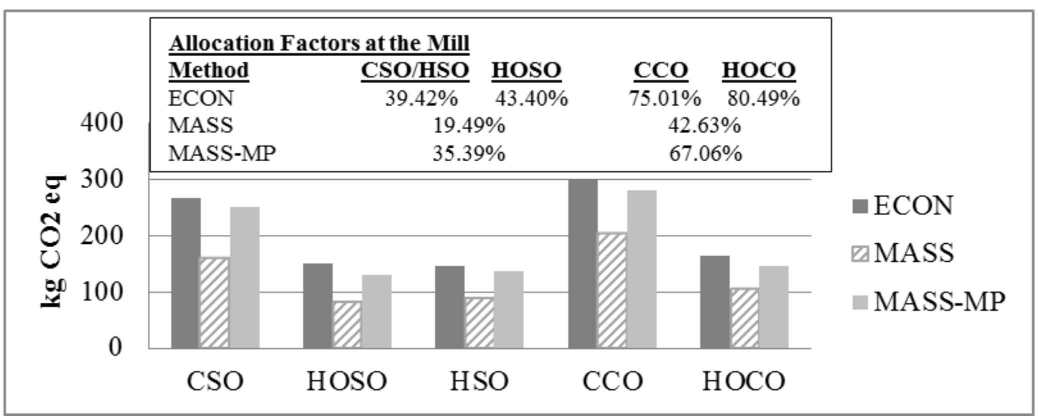

Figure 3: Allocation Sensitivity: CCP Impacts for Scenario 1 using Economic, Mass (Meal/Oil), and Mass-MP (Meal Protein/Oil) Allocation Methods. Allocation factors at the mill are included for reference.

3.5. Spent Oil Sensitivity

As demand for biodiesel increases, the potential outlets for spent oil for the restaurant owner may become more plentiful and profitable. While the current scenario assumes no burdens or credits for the spent oil, this sensitivity analysis assumes the restaurant receives \$0.11/kg (i.e. \$0.05/lb) for the spent oil on a dry basis. It is modelled as an economically equivalent quantity of crude (i.e. extracted) conventional oil that is avoided from the overall burden of the fryer system. In other words, roughly 0.1 kg crude soybean oil is avoided per 1 kg spent soybean oil produced. No packaging materials are considered with respect to spent oil collection.

For scenario 1, overall impacts are reduced 6%-10% for conventional oils depending on the impact category, and 5.5%-8.1% for high oleic oils, with the reduction being 15% more for conventional oils than for high oleic oils for any specific impact category. For scenario 2, overall impacts are reduced 4.5%-5.9% for conventional oils but only 1.7%-2.5% for high oleic and hydrogenated oils. Figure 4 highlights the CCP impacts with and without the spent oil credit for conventional and high oleic soybean oil.

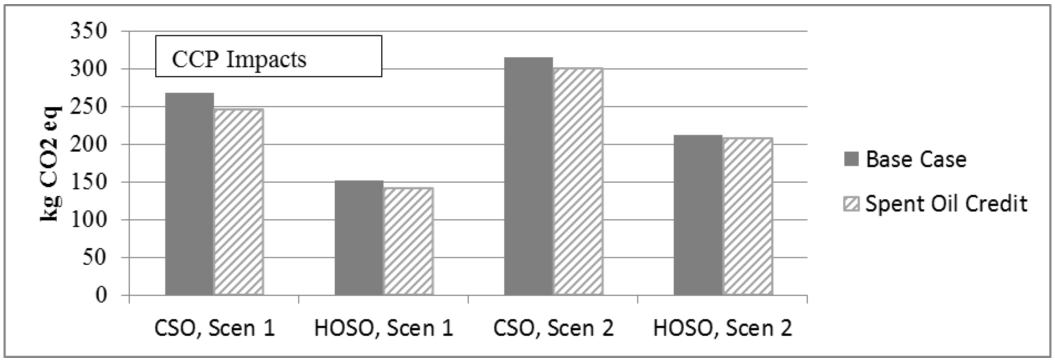

Figure 4: Spent Oil Sensitivity: CCP impacts for CSO and HOSO for both Scenarios 1 and 2 for the base case and assuming a $0.11/kg spent oil credit.

4. Discussion

When comparing among the environmental impacts evaluated in this study of the three soybean oil options, oil performance and restaurant behavior account for the majority of the differences in impacts, while mill co-product allocation, spent oil allocation, and oil processing have more subtle impacts. The increased stability of both the high oleic and the hydrogenated oils allows for lower oil use rates and fewer washes at the restaurant. The amount of food cooked during the day will affect the oil absorption rate, with increased absorption dampening the environmental benefits of the HOSO and HSO since the benefits of increased stability are not realized for absorbed oil. Impacts can be reduced as much as 48% based on the increased stability compared to CSO. As absorption rates increase from 4% to10% the differentiation among CSO and HSO environmental impacts is reduced by 20%. The allocation methodology can have significant impacts to the overall environmental burdens of the soybean oil options. Mass allocation results in impacts as much as 40% lower than economic allocation, but has less importance when differentiating among the oils types. For instance, CCP for HOSO is 44% lower than CSO using economic allocation and 48% lower using mass allocation. The environmental impacts of increased processing required for hydrogenation for HSO counterbalances the higher economic allocation impacts of HOSO, resulting in 4% to 6% lower impacts than HOSO for CCP and EP, but 3%-4% higher impacts for NRE and AP. For MASS and MASS-MP allocation methods, HSO consistently has higher impacts than HOSO. Soybean yields, farming practices, electricity mix assumptions, and oil transport distance do not serve to differentiate among these options since they are the same for each. Similar conclusions can be made when comparing CCO to HOCO.

However, when comparing different types of oils, such as CSO to CCO, or HOSO to HOCO, other facets of the oil supply chain become important. Farming fertilizer use is significantly higher for canola, leading to increased impacts for canola oil, particularly for AP and EP. As legumes, soybeans have N-fixating capabilities which significantly reduce the amount of N-based fertilizer required. This, in turn, reduces CCP, AP, and EP due to reduced fertilizer production impacts and field emissions. Crop yield favors soybeans as canola is shown to require as much as 50% more land under Scenario 1 assumptions. Although modeled with similar emissions per hectare, nitrous oxide field emissions have a substantial impact on the overall climate change and the relative impacts for soybean and canola oils. These emissions will likely differ by region for each crop, local field conditions, farming practice, etc., and have perhaps more than 100% uncertainty based on the USB report where N_2O emissions increased two-fold as they moved from calculated emissions to a site specific measurement (Omni-Tech, 2010). Finally, transport of the oil becomes more important. Although only 1%-13% of the total CCP impact for the cities evaluated, the transport model assumptions can represents more than 60% of the relative difference between soybean and canola oils for CCP. Transport accounts for roughly 30% of the difference between soybean oil and canola oil for NRE with little to no impact on AP, EP, and Land Use.

5. Conclusion

Overall, oil functionality was shown to be a key driver and a new handle on reducing environmental impacts for frying oil applications. Improvements to oil stability facilitated by high oleic seeds reduces oil use requirements and restaurant washing requirements. Increases in economic allocation factors for the high oleic oils do not dampen the impact reductions significantly. Restaurant behavior in terms of food throughput and oil replacement frequency as well as the food's oil absorption rate may significantly influence the extent of benefit. Similar environmental impacts are seen for HSO in comparison to HOSO as they both provide similar increased oil stability, but the hydrogenated oil alternative is compromised with trans-fats and higher saturated fat content. While yield, allocation methodology, transportation assumptions, and spent oil allocation all have the potential to affect the overall magnitude of frying oil environmental impacts, and may influence comparison across oils from different seeds, they have limited impact when comparing high oleic and conventional oils from the same feedstock. In addition to having nutritional benefits, high oleic soybean oil has been shown to reduce all environmental impacts evaluated in this study relative to conventional soybean oil for fryer use in the U.S.

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Creating coherent life cycle databases for ecodesign and product declaration of agroindustrial products: how to deal with contradictory methodological requirements

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ABSTRACT
Existing guidelines and standards for creating LCI databases provide partly contradictory requirements which lead to initiatives that aim on harmonization. As the harmonization is still ongoing, this challenges current database projects to find a scientifically sound and applicable way to establish coherent datasets. We present a four-step approach to deal with this challenge. Based on our experiences in the two ongoing projects ACYVIA (Analyse de CYcle de Vie dans les Industries Agro-alimentaires) and WFLDB (World Food LCA Database) we draw the following conclusions: it has been shown that by following the proposed approach, most contradictory advices from different guidelines do not appear because the number of relevant guidelines can be reduced. Creating a database that allows different methodological decisions can be achieved by clearly defining and reporting all methodological decisions that are followed. For existing contradictory requirements, decision criteria are presented that can be taken into consideration to decide for one specific requirement.

Keywords: LCI database, agri-food sector, methodological guidelines, harmonization

1. Introduction

Agricultural production systems and the processing of agricultural raw materials to food products contribute significantly to several environmental impacts like global warming, eutrophication and acidification (Pardo and Zufia 2012; Ruviaro et al. 2012; Saarinen et al. 2012). Emissions from agricultural production systems show a high temporal and spatial variability which is a reason for a high variability of environmental impacts of these systems (Mouron et al. 2006; Roy et al. 2009; Nemecek et al. 2012; Rees et al. 2013). These facts together with an increasing public interest enforce the demand for LCI data in the agri-food sector in companies, science and governments in the last years. Various guidelines exist (see Table **1**) with partly contradictory requirements which causes confusion (Finkbeiner 2014). A recent review of such reference methods conclude that flexibility with respect to methodological standards is more common than prescriptive requirements are (Pelletier et al. 2014) In this context, several initiatives and projects deal with the creation on LCI databases that are either focused on the agri-food sector or cross-sectorial including agri-food related content, e.g. ACYVIA (Bosque et al. 2012), Agri-BALYSE® (Koch and Salou 2013), Asian Agri-Food database (Hayashi 2013), Australian LCI Database initiative (ALCAS 2014), Base IMPACTS® (ADEME 2014), Chilean Food and Agriculture LCA database (Emhart et al. 2013), ecoinvent (Weidema et al. 2013), ELCD (JRC 2014), World Food LCA database (Lansche et al. 2013).

This paper wants to start a discussion on the question how one can deal with the situation of existing guidelines and standards with contradictory requirements when creating an LCI database. The focus is on LCI modelling and the ideas presented are not final solutions but aim on being a starting point for further discussions. Basically, three steps are presented:

1) Categorizing the database to select the appropriate standard, guideline or tool for the purpose of the database to avoid contradictions
2) Showing an example for dealing with the requirement that a database should be applicable for different purposes
3) Developing basic principles on how to deal with remaining contradictions

Table 1. Non exhaustive list of existing guidelines and standards.

Short Title	Full title of the guideline or standard	Reference
BPX 30-323-0	Environmental communication on mass market products — Part 0: General principles and methodological framework	Afnor (2011)
PAS 2050:2011	The Guide to PAS 2050:2011: How to carbon footprint your products, identify hotspots and reduce emissions in your supply chain	BSI (2011)
PEF Guide	Product Environmental Footprint (PEF) Guide, Annex II to the Recommendations of the Commission of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organizations	EC (2013)
Envifood protocol	Environmental Assessment of Food and Drink Protocol	Envifood (2013)
MTT Guidelines	Guidelines for the assessment of the life cycle greenhouse gas emissions of food	Hartikainen et al (2012)
IDF Guide	A common carbon footprint approach for dairy – The IDF guide to standard lifecycle assessment methodology for the dairy sector	IDF (2010)
IPCC Guidelines	Guidelines fo National Greenhouse Gas Inventories -Agriculture, Forestry and other Land Use.	IPCC (2006)
ISO 14025:2006	Environmental labels and declarations - Type III environmental declarations - Principles and procedures	ISO (2006a)
ISO 14040:2006	Environmental management - Life cycle assessment - Principles and framework	ISO (2006b)
ISO 14044:2006	Environmental management - Life cycle assessment – Requirements and guidelines	ISO (2006c)
ISO 14067:2013	Carbon footprint of products—requirements and guidelines for quantification and communication.	ISO (2013)
ILCD Handbook	International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance	JRC (2010)
Shonan Guidance Principles	Global Guidance Principles for Life Cycle Assessment Databases , A basis for greener processes and products	UNEP/SETAC (2011)
Ecoinvent data quality guidelines	Overview and methodology. Data quality guideline for the ecoinvent database version 3	Weidema et al (2013)

2. Methods

The following methodological procedure is a proposition on how a coherent database could be created given the various guidelines and methodological recommendations as illustrated in Table **1** above. We suggest a procedure with the following main steps:

- Step 1: Categorizing the database as "general database" or "specific database". For categorizing a database we propose to use specifications for the geography, application, and sector that are addressed given in Table **2**.
- Step 2: Identify the most relevant guidelines (from Table **1**) related to the database.
- Step 3: Identify the methodological options that are crucial for the database. Options for LCI occur e.g. for system boundary choice, direct emission modeling, allocation methods, end-of-life modeling, data source choices and the kind of dataset documentation.
- Step 4: Decide which options to use in order to meet the criteria according to Table **2**.

This four-step procedure is applied to two ongoing database projects that are:

- WFLDB (World Food LCA Database): This project is developing datasets for selected agricultural primary products as well as food and beverage products produced in the most relevant countries that supply the global market.
- ACYVIA (Analyse de CYcle de Vie dans les Industries Agro-alimentaires): This project addresses environmental product declaration of food transformation processes at national-level in France.

Table 2. Categorizing food databases

Criteria	**General database**	**Specific database**
Geographical specification	Global, multi-national	National, regional
Application addressed	Ecodesign and Environmental product declaration (EPD)	Ecodesign or Environmental product declaration (EPD)
Sectorial specification	Agriculture and food industry	Agriculture or food industry

3. Results

Categorizing databases

The two database projects WFLDB and ACYVIA can be clearly categorized with as "General database" and "Specific database", respectively (Table **3**). Table 2 shows also that the two project differ very much in the order of guidelines that are most relevant for each project. For ACYVIA the BPX guidelines are of the highest importance defining methods for LCI modelling, system boundaries, allocation, end-of-live modelling whereas the ILCD entry-level is of importance regarding the method for data quality assessment and the selection of external reviewers. As a consequence, in case of the ACYVIA database practically no methodological options are left since BPX defines them all for EPD in France. In contrast, for WFLDB due to the wide range of geographical, sectorial applications a number of methodological decisions according to ISO 14044/ 44 have to be taken. In practice this means that for each methodological issue one option has to be chosen. Such choices need to be described in the documentation of the database. But whatever option is chosen, it might be that for a certain database user and for certain applications this methodological option is not the one that suits well. Therefore we model a methodological option in a reversible way, that means, the user will have the opportunity to calculate backwards and to apply another methodological option that fits to the desired application. This is e.g. the case when economic allocation is applied.

In the following we will illustrate for the case of modelling "heavy metal uptake by crops" what is meant by reverse modelling:

Table 3. Categorizing WFLDB and ACYVIA database and associated relevant guidelines

	WFLDB **General database**	**ACYVIA** **Specific database**
Geographical specification	Global	National
Application addressed	Ecodesign and EPD	EPD
Sectorial specification	Agriculture and food industry	Food industry
Guidelines (order of importance)	1. ISO 14040/ 44 2. ILCD handbook 3. ENVIFOOD 4. Others	1. BPX 30-323-0 2. ILCD entry-level 3. ISO 14040/ 44 4. Others

Reverse modelling of heavy metal uptake by crops

In crop production heavy metals (e.g. Cadmium) will be imported to the field by inputs such as mineral fertilizers. On the field the plant is taking up nutrients but also heavy metals that will be exported from the field with the harvested crop. In case the whole life cycle (i.e. from cradle to grave) is assessed the amount of heavy metal exported by the crop is of interest since this might cause toxicological problems at another place (e.g. waste water treatment after consumption and digestion). But if the LCA addresses only the crop production on the field (i.e. cradle to gate) the uptake of heavy metal could lead to unrealistic “credits” and therefore want to be excluded from the assessment. We suggest to model heavy metal in that way that the uptake to the plant can be set to zero, if needed.

4. Discussion

We proposed a first approach how one can deal with the situation of guidelines and standards with contradictory requirements when creating an LCI database. The three criteria (geography, application, economic sector) for categorizing databases have been sufficient for the two projects WFLDB and ACYVIA but its sufficiency and applicability need to be proved in practice with other databases.

If contradictions remain, we propose to develop a hierarchy of basic principles that support to make appropriate methodological decisions in respect to LCI modelling. Such criteria can be:

- scientific nature of the requirement
- internal consistency of the database
- acceptance by stakeholders

The ideas presented have to be further developed and tested more comprehensively in practice.

5. Conclusion

By categorizing databases, relevant guidelines can be selected. This helps to identify the relevant methodological options. By following this approach, most contradictory advices from different guidelines do not appear because the number of relevant guidelines can be reduced for each individual database.

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Biogas or feed – different pathways for selected food industry residues from a greenhouse gas perspective

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ABSTRACT

In this paper, GHG emissions of biogas systems based on food industries residues are analysed according to the method described in the ISO-standard for life cycle assessment. Furthermore, two perspectives are included, one where the residues are not utilised for other purposes today and one where the residues are utilised as animal feed, here presented as systems expansion. The results show that all residues studied are well suited for biogas production if there is no demand for them as animal feed today. Otherwise, it is often more efficient to grow dedicated biogas crops directly.

Keywords: biogas, food industry residues, feed, LCA

1. Introduction

Production of food results in various kinds of food industry residues that can be utilized in different ways depending on their individual features and demand from other market sectors. Often, such residues are used as feed but in some regions this demand decreases due to changes in livestock production. Also, there is a growing market for biofuels where such feedstock could be utilized as well. Recently, the use of industrial residues for the production of biogas as transportation fuel has also attracted a great deal of interest since such production does not compete directly with the use of agricultural land. However, only a limited number of studies quantifying greenhouse gas (GHG) emissions from such biogas systems have been performed. For comparison, the environmental performance of other biofuels such as ethanol and biodiesel as well as biogas from waste and dedicated energy crops has been investigated in several studies (see e.g. Börjesson and Tufvesson, 2011, JRC, 2007, Kendell and Chang, 2009 and Reijnders and Huijbregts, 2008). Also, if the demand for feed remains and such feedstock are used for biogas production, they must be replaced by other animal feed. This may shift the environmental burden from one system to another and needs to be included in a system analysis.

In this paper, which is mainly based on the findings presented in Tufvesson et al. (2013), GHG emissions from different biogas system, where biogas is utilized as a vehicle fuel produced from industrial residues, are analyzed and compared from a life-cycle perspective including different utilization pathways for the residues. Residues included are distiller's waste, rapeseed cake, whey permeate, fodder milk and bakery residues. Also, calculated GHG emissions from biogas systems based on food industry residues are compared to corresponding emissions from biogas systems based on dedicated energy crops.

2. Methods

The analysis is based on the principles described in the ISO standard 140 44 (ISO, 2006). The functional unit is 1 MJ of upgraded and compressed biogas at the biogas plant. Included life cycle emissions of GHG are carbon dioxide (CO_2) of fossil origin, methane (CH_4) and nitrous oxide (N_2O). Characterization factors are set to 1, 25 and 298 g CO_2-ekq./g CO_2 respectively (IPCC, 2006a).

Data is presented for two scenarios; no allocation and system expansion. In the scenario with no allocation all feedstock are considered as residues and no emissions are allocated to them before entering the biogas system. In addition to the upgraded and compressed biogas, all biogas systems also generate digestate which is applied to arable land as a fertilizer. Thus, GHG emissions could be allocated between biogas and digestate. However, this digestate has no clearly defined value today and due to its low dry matter content, it is not considered to have a heating value either. Thus, all GHG are allocated to the biogas produced (Tufvesson et al., 2013). When system expansion is applied, food industry residues are assumed to be replaced with soymeal and barley as animal feed. Also, in the case with system expansion, the digestate is assumed to replace mineral fertilizer.

Inventory data for biogas production and upgrading are collected to represent average new technology that is commercially available today. Inventory data for generation of electricity used in the biogas system, production of mineral fertilizers and animal feed has been chosen to represent average products used in Sweden today.

2.2. Investigated substrates

The substrates investigated in this paper have different features which affect the production of biogas and digestate as well as the amount of mineral fertilizers and animal feed that can be replaced. Also, the same kind of food industry residue could have different features depending on the original raw material and how it has been processed. The features assumed in this study are presented in Table 1. Further background on assumptions made can be found in Tufvesson et al. (2013).

Table 1. Features of analyzed substrates (Tufvesson et al., 2013).

Substrate	DM[a]	Plant available nutrients (% of DM)			Methane yield	Corresponding feed (kg DM/kg DM substrate)	
	(%)	N	P	K	Nm^3/tonne DM	soy meal	barley
Distiller's waste	8,0	4.0	0.9	1.1	302	0.4	0.6
Rapeseed cake	91	5.0	1.2	0.7	422	0.7	0.3
Whey permeate	5,0	0.5	0.8	2.5	309	0	1.1
Fodder milk	9,0	3.9	0.8	1.0	472	0.4	1.5
Bakery residues	61	3.7	0.2	0.4	304	0	1.0

[a] Dry matter

2.3. Analyzed biogas systems

It is assumed that all food industry residues are transported by truck to a centralized co-digestion plant where biogas is produced, upgraded and compressed. Corresponding to the system boundaries applied by Börjesson *et al.* (2010) further transportation of compressed biogas is not included. The transportation distance is set to 10 km for liquid substrates and 30 km for rape seed cake and bakery residues and the energy input for transportation is set to 1,1 MJ/tonne*km (Börjesson and Berglund, 2006).

Electricity consumption at the biogas plant is set to 30.2 MJ/tonne of substrate in addition to the electricity used for upgrading and compression which is set to 1.6 and 1.3 MJ/Nm^3 of biogas respectively. Emissions are based on average Swedish power production and corresponds to 11 g CO_2-eqv./MJ. Also, it is assumed that the biogas plant use 93.6 MJ of biogas to produce process heat (Lantz et al., 2009; Tufvesson et al., 2013).

The methane leakage is set to 1.5% of the biogas produced including biogas production, upgrading and digestate storage at the plant representing a modern biogas plant. For comparison, the average methane losses from Swedish co-digestion plants and upgrading plants were 1.8% and 1.4% respectively in 2012 (Avfall Sverige, 2012). However, there are also biogas plants with state of the art technology where methane losses could be 0.5% or lower.

The digestate produced is transported 10 km by truck, stored in covered concrete tanks and spread on arable land by tractor. Based on IPCC (2006b) it is assumed that there will be no direct emissions of N_2O from the digestate storages. Emissions of NH_3 from the storage tanks are set to 1 % of the total amount of nitrogen in the digestate which also cause indirect emission of N_2O (IPC, 2006b). The amount of indirect N_2O is calculated based on equation 1.

$$N_2O = N*EF*(44/28) \quad \text{Eq. 1}$$

N_2O = indirect emissions of nitrous oxide
N = emissions of nitrogen as NH_3-N (kg N)
EF = emission factor, which is here set to 0.01 kg N_2O-N/kg NH_3-N

In the system expansion, indirect emissions of N_2O from spreading of the digestate are included as well. These emissions are calculated according to Equation 1 assuming that 5 % of the total amount of ammonium nitrogen in the digestate is lost at NH_3.This assumption requires good spreading techniques as well as appropriate weather conditions (Tufvesson et al., 2013).

The amount of mineral fertilizers that could be replaced by digestate is presented in Table 1. Based on the market situation in Sweden, it is assumed that 30% of the mineral nitrogen is produced with catalytic N_2O reduction resulting in average emissions of 6.7 kg CO_2-eqv./kg N. For P and K, emissions are set to 3.2 and 0.5 g CO_2-eqv./kg respectively (Tufvesson et al., 2013).

In addition to the nutrients presented in Table 1, digestate also contains carbon compounds of which some will form stable humus in the soil. In this analysis it is assumed that 40% of the dry matter in the digestate is carbon and 10% of this carbon is conservatively estimated to form long-term stable soil organic matter. A more comprehensive background for this assumption is presented in Tufvesson et al. (2013).

The assumed amount of crops needed to replace the various substrates as animal feed is presented in Table 1. GHG emissions from production of soybean meal and barley are set to 980 and 450 g CO_2-eqv./kg DM respectively (Flysjö et al., 2008). Emissions are calculated assuming average cultivation conditions, including direct land-use change, but not any indirect land-use change.

3. Results

In Figure 1 the contribution to global warming potential is shown for the biogas systems analyzed. The results vary considerably between the case with no allocation and the case with system expansion. However, for all the systems studied the contribution to global warming was lower than for petrol and diesel. In Table 2 it is clearly seen that the parameters that contribute most to the global warming potential are the replacement of mineral fertilizers and animal feed in the system expansion.

In figure 2, GHG emissions from biogas systems based on food industry residues, including system expansion, are compared to biogas systems based on cultivated crops, dedicated for biogas production. Although all biogas systems reduce GHG emissions compared to petrol and diesel, it is clear that the reduction is higher when dedicated energy crops are utilized.

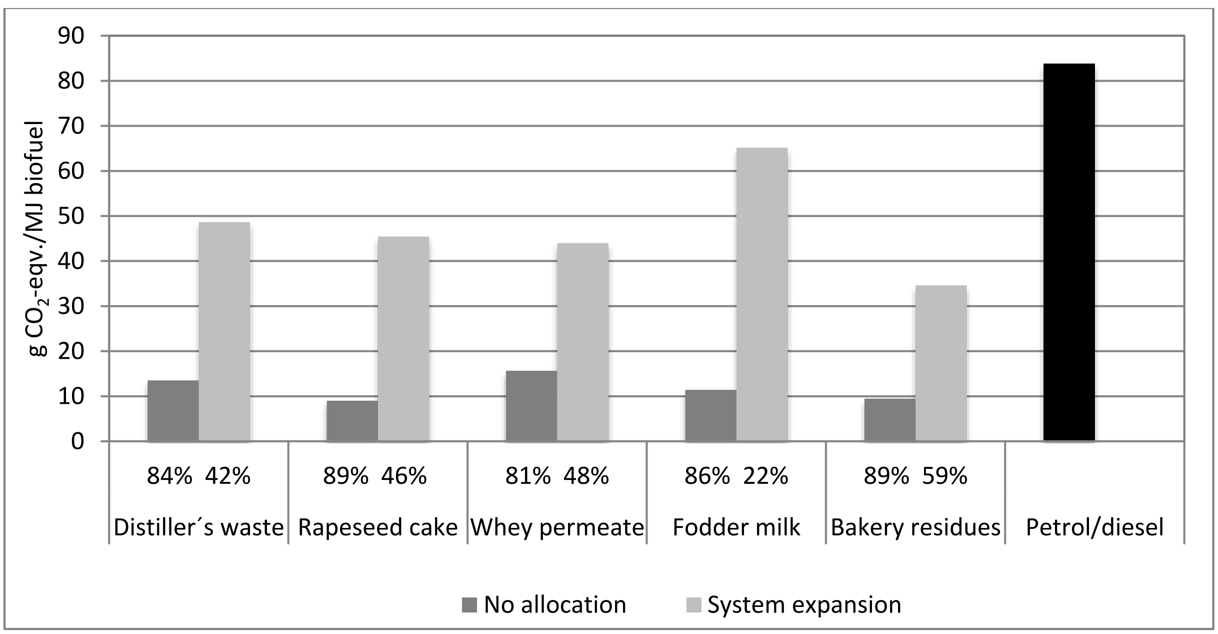

Figure 1. Contribution to global warming potential from biogas based on industrial residues (taken from Tufvesson et al., 2013). The reference for petrol and diesel is set to 83,8 g CO_2-eqv./MJ (European Commission, 2009).

Table 2. The total contribution and most important parameters contributing to global warming potential when no allocation and system expansion is applied (taken from Tufvesson et al., 2013).

Substrate	**Global warming potential (g CO_2-eqv./MJ biogas)**							
	Transport[a]	**Process energy**	**Methane leakage**	**Digestate**[a]	**Total no allocation**	**Mineral fertilizer**	**Animal feed**	**Total system expansion**
Distiller's waste	1.1	1.5	7.5	3.5	13.6	-23	58	49
Rapeseed cake	0.2	0.9	7.5	0.4	9.0	-19	55	45
Whey permeate	1.5	1.4	7.5	4.9	15.3	-8	38	44
Fodder milk	0.6	1.1	7.5	2.0	11.2	-12	66	65
Bakery residues	0.4	1.0	7.5	0.7	9.6	-21	46	35

[a] Transport of substrate

[a] Storage, transport and spreading of digestate

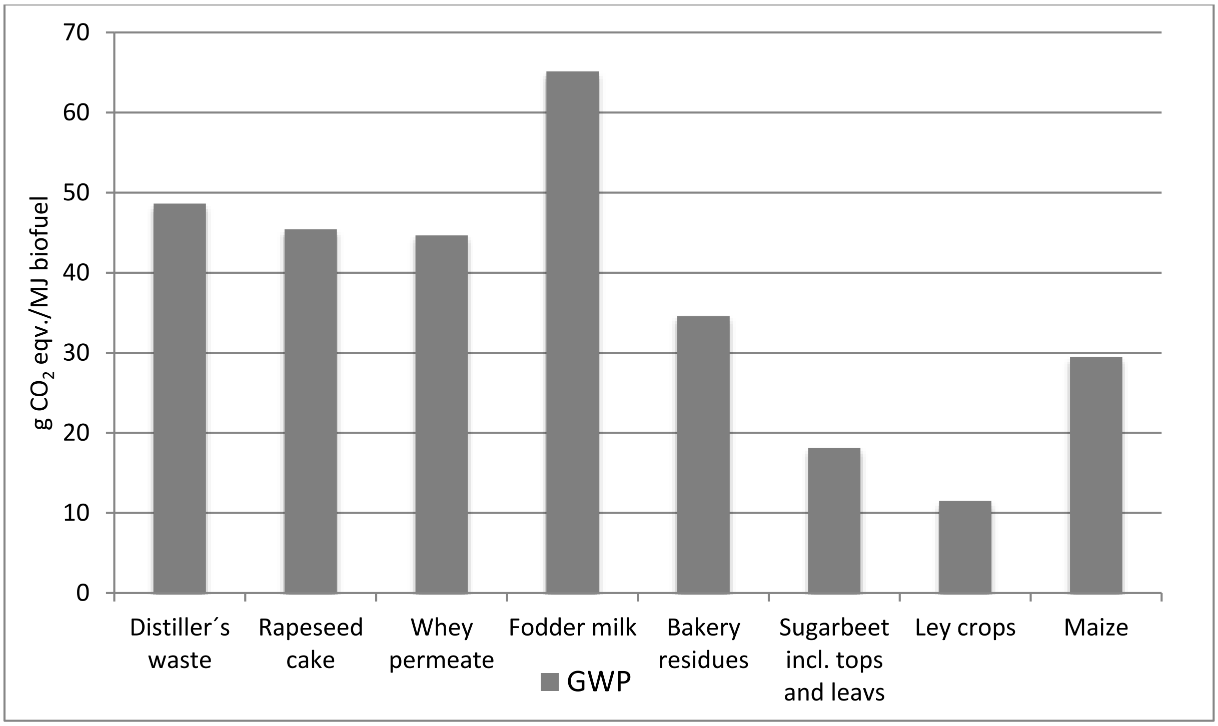

Figure 2. Contribution to global warming potential when system expansion is applied for biogas from industrial residues and dedicated crops (unfertilized grassland as reference land-use) taken from Börjesson et al., (2010).

4. Discussion

Many kinds of food industry residues are well suited for biogas production since they contain organic matter that could relatively easy be degraded in an anaerobic process. The residues are also concentrated to few industrial sites and could easily be transported to a biogas plant in the vicinity of the site or on the site. When the biogas produced is utilized to replace fossil fuels, all analyzed biogas systems also reduce GHG emissions compared to petrol and diesel. This approach is also encouraged by the debate on iLUC factors and the promotion of using residues and waste for biofuel production which e.g. is the case in the EU's Renewable Energy Directive (European Commission, 2009). However, this is a simplified view based on a limited systems perspective since it does not include the potential alternative utilization of the residues. If there is a demand for the residues as feed, their utilization as biogas feedstock will be compensated for by, e.g. new feed crop production. According to the ISO-standard of LCA (ISO, 2006), system expansion should also be applied when possible to cover all relevant indirect effects influencing the result.

As presented in this study, the utilization of food industry residues as feed, if there is such a market, will thus result in the highest reduction of GHG. Food industries considering the best way how to handle their organic residues from a greenhouse gas perspective should thus closely evaluate the possibility to utilize such residues as feed. However, if there is no such market, biogas production is a suitable alternative.

5. Conclusion

Food industry residues analysed in this paper are well suited for biogas production and the reduction of greenhouse gas emissions will be substantial compared with fossil vehicle fuels. One important prerequisite is, however, that the industrial residues cannot be utilized as animal feed. Otherwise, the benefits will be significantly reduced due to additional production of feed crops. If so, it could be more efficient to grow dedicated biogas crops and continue to utilize the industrial residues as animal feed.

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Environmental benefits of compost use on land through LCA – a review of the current gaps

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ABSTRACT

The use of biowaste compost on land can have beneficial effects on the plant–soil system. While the environmental impacts associated with compost production have been successfully assessed in previous studies, the assessment of the benefits of compost on plant and soil has been only partially included in few published works. In the present study, we reviewed the recent progresses made in the quantification of the effects associated to biowaste compost use on land by using life cycle assessment (LCA). Different research efforts are required for a full assessment of the potential benefits, apart from nutrient supply and carbon sequestration; additional impact categories – dealing with phosphorus resources, biodiversity, soil losses, and water depletion – may be needed for a comprehensive assessment of compost application. Several of the natural mechanisms identified and the LCA procedures discussed in the paper could be extensible to other organic fertilizers and compost from other feedstocks.

Keywords: sustainable agriculture, organic fertilizer, biowaste, soil organic matter, soil quality, biodiversity, carbon sequestration

1. Introduction

There is increasing concern about soil interrelated environmental problems such as soil degradation, desertification, erosion, and loss of fertility (European Commission 2006). These problems are partially consequence of the decline in organic matter content in soils. An estimated 45% of European soils have low soil organic matter (SOM) content, principally in southern regions (European Commission 2006). A parallel concern is the massive generation of organic waste by human activities in urban areas. Composting is one of the best-known and well-established processes to recycle organic waste. Composting allows the stabilization and sanitation of organic waste through accelerated aerobic decomposition under controlled conditions, resulting in a product called compost.

Several studies indicate that the use of compost on land may improve several plant and soil parameters, thereby making compost an interesting option for soil restoration purposes, while taking advantage of its fertilizer properties. On the other hand, the application of compost may also potentially result in environmental and agronomic drawbacks, such as gaseous and leachate emissions, and increase in salt and heavy metals content in soil (Hargreaves et al. 2008). Nevertheless, these issues are in general directly associated with the quality of the final compost and agronomic management.

Life Cycle Assessment (LCA) was promoted in different European directives as a robust quantitative tool and has been widely used for the environmental assessment of waste and agricultural sectors. The negative environmental impacts associated with compost production and transport, particularly in the case of compost produced from municipal solid waste, have been successfully assessed in previous studies (ROU 2007; Boldrin et al. 2009; Martínez-Blanco et al. 2010; Colón et al. 2012). However, assessment of most of the potential environmental benefits of compost on plant and soil has not been carried out yet. Several recent studies address the inclusion of compost benefits in a qualitative manner, recommending that further research should be undertaken on the subject (Boldrin et al. 2009; Favoino and Hogg 2008; Hansen et al. 2006; Martínez-Blanco et al. 2011). Nevertheless, carbon sequestration and nutrient supply are, to date, the only environmental benefits taken into account in these studies. Because of the modeling complexity, ROU (2007) is, to our knowledge, the only study that attempted including most of the abovementioned benefits within LCA of two Australian case studies. The results were however only presented at the inventory stage.

The main goal of this review paper is to quantitatively address LCA modeling of the positive potential effects traditionally associated with land application of biowaste compost produced from organic municipal solid waste and garden waste. Here we have focused only on the implications of compost application to the soil and plant without considering the full life cycle (i.e. production process and transport are not discussed here).

2. Methodology

A comprehensive review of the literature dealing with the potential benefits of compost application and the current situation of the inclusion of each of these benefits in LCA studies was carried out. First, the most relevant benefits of compost on soil properties and plant growth were identified and the inventory data was collected. Subsequently, 90 articles (including both reviews and case studies) published later than 1990 were selected. Although similar environmental and agronomical benefits could be observed in compost produced from other types of feedstock and in other organic fertilizers, in this review field studies considering compost from organic municipal solid waste and green waste (from now on called biowaste) were taken into account when possible. The potential benefits were grouped into nine categories (Table 1). According to the literature review, the benefits were classified into short-term (1 year), mid-term (1–10 years), and long-term (10–100 years), depending on the time perspective of the agronomic effects.

Later on, the potential benefits studied were reviewed, through an LCA perspective, according to: (1) the existing evidences for the effects on soil, plant, environment, farmer or harvest and the main factors affecting the results for each of them; (2) the possibility of quantification of the substituted or saved process (i.e. the availability of data that can be later included in an inventory); and, finally, (3) the current availability of tools for their inclusion in LCA, together with the current status of new assessment methodologies.

3. Results

The following two sections provide an overview of the results. Occurrence of individual benefits is discussed in the first part, while the second part deals with quantification of the benefits in a life cycle perspective.

3.1. Compost potential benefits

An outline of the literature review dealing with the nine potential benefits resulting from compost application is provided in Table 1. The full review is available in Martínez-Blanco et al. (2013) where, for each of the agronomic benefits, a discussion of the main factors affecting the performance of individual benefits, the degree of proof and the range of the benefits measured were included.

Regarding the supply of plant nutrients, carbon sequestration, soil erosion and soil workability, the positive effects of compost application were demonstrated in most of the reviewed studies, and their magnitude was quantifiable. Although we were also able to state the magnitude of the effect, for the following three benefits the share of studies with non-significant results was relevant: crop nutritional quality was not relevantly different for a third of the case studies included; for crop yield, more than 60% of the case studies did not report differences when compost was applied; and finally, non-significant benefits were detected for soil moisture content for low rates of compost. These were also the benefits with higher disparity in the measured effects among the results.

For the benefits pest and disease suppression and crop nutritional quality, although they were proved, it was not feasible to summarize the benefit in a unique data range. These benefits involve several concurring indicators at the same time and the intensity of the effect is different for each of them due to several factors. Regarding weed suppression, this effect was not proved when compost is used as a soil amendment. Finally, data regarding effects of compost application on soil biological properties and biodiversity are scarce and restricted to microorganisms. Table 1 shows the results of the review for three of the most used microbial indicators.

On average, positive effects due to compost application were found for all the potential benefits, except for weed suppression. Benefits in the long-term were only reported for nutrient supply, carbon sequestration, soil biodiversity and soil workability, whereas for the other potential benefits only mid- or short-term data were found. In addition, quantification of the potential benefits yielded broad ranges in most of the cases.

During literature review the variables having the largest influence on the magnitude of compost benefits were identified. The original feedstock material, management of the composting process, compost maturity, and crop management are some of the main factors that determine the occurrence of environmental and agronomic benefits. For instance, Boldrin et al. (2009) reported that the typical contents of nutrients in biowaste compost can vary depending on the initial raw waste material. Susceptibility of these nutrients to mineralization and release might depend on the degree of stability and/or maturity of the compost as well as on the prevailing climatic conditions due to the large influence of temperature and moisture in decomposition and nutrient release (Sikora and Szmidt 2004).

Regarding the impacts of compost on soil moisture, workability and erosion, several authors reported large positive effects with high-rate compost application on soils with initially low SOC content. Compost quality is the most important factor determining the impacts on soil biological properties and biodiversity, together with the dose applied (Hargreaves et al. 2008; Diacono and Montemurro 2010). Increases in crop nutritional quality when compost is employed largely depend on crop management and climate conditions (e.g. better results were observed when a lag of time between compost application and crop existed and nutrient mineralization rates tend to be higher in warm climates).

3.2. Quantification of benefits

Depending on the nutrient content and availability of the compost, the use of mineral fertilizers can be avoided and therefore their industrial production and transport. Final utilization efficiencies for N, P and K are in the order of 20–60% for N, 90–100% for P, and 100% for K (Boldrin et al. 2009). Using the nutrient contents presented in Table 1, the potential amount of inorganic fertilizers replaced may be within the range of 1–13 kg of N, 1–5 kg of P, and 5–14 kg of K per ton of compost applied. A life cycle inventory (LCI) for fertilizer production include use of materials and energy, and emissions to different compartments, which would typically result in potential impacts on Resource Depletion, Global Warming, Human- and Ecotoxicity, and Eutrophication impact categories (Figure 1). Inventory datasets for N-P-K fertilizers are reported in different sources. In addition, the use of a renewable P source rather than inorganic non-renewable supply is of great importance and this reduced raw resource consumption might be quantified during impact quantification.

The amount of C sequestered into soil by compost application, can be translated into saved CO_2 emissions by using a conversion factor of 44/12, based on molar relation, and then entered into the LCI. A time frame of 100 years is considered to be relevant for estimating contributions to Global Warming (Favoino and Hogg 2008). Boldrin et al. (2009) reported that the benefits from C retained in soil 100 years after the addition of biowaste compost is between 2 and 79 kg CO_2-eq. t^{-1}. Higher values, 279 kg CO_2-eq. t^{-1}, were reported by ICF (2005). Most likely, this large variability is due to the synergetic effect of the different environmental and site-specific factors, meaning that estimations should be done on a case-to-case basis.

When the application of compost results in pest suppressive effects, the use of pesticides can be reduced or avoided. However, compost benefits on plant health are so case-specific that it is not possible to provide any general figures for the amount and the type of pesticides saved. The avoided use can be credited to the system as an environmental saving associated to both the avoided production/transportation and to the avoided release of these products to the environment. Inventory data covering emissions modelling, production and transport of pesticides can be found in different databases such as PestLCI, Ecoinvent, and GEMIS. Potential environment impacts from production/transportation and use of pesticides can be assessed using existing impact categories, the most relevant being the Toxicity categories, both Human- and Ecotoxicity (Figure 1).

Increased yield as a consequence of compost application could result in avoided additional agricultural production, and thus all the associated environmental burdens. If arable land is not constrained, the benefit is linked to theoretical avoided use of material and energy needed for the crop production (of the yield increased). In the most likely regime of constrained arable land, the increased yield would have an effect on both intensification and expansion of agricultural production, and ultimately will prevent indirect land use changes (ILUCs), which are for instance a major source of GHG emissions (Thamsiriroj and Murphy 2010). Depending on the specific area and crop, most of the impact categories are influenced when agricultural production is involved (Figure 1).

Table 1. Summary of the potential benefits of compost use-on-land in the short-, mid- and long-term retrieved from the literature review (adapted from Martínez-Blanco et al. 2013).

Benefit	Indicator (Unit)	Short-term (<1 yr)		mid-term (<10yr)		long-term (<100yr)	
		Min.	Max.	Min.	Max.	Min.	Max.
Nutrient supply	N mineralized (% of N applied)	5	22	40	50	20	60
	P mineralized (% of P applied)	35	38	90	100	90	100
	K mineralized (% of K applied)	75	80	100		100	
Carbon sequestration	C sequestered in soil (% of C applied)	40	53		30	2	16
Weed, pest and disease suppression	Weed suppression (-)	ns	ns	-	-	-	-
	Pest and disease suppression (-)	nad	nad	-	-	-	-
Crop yield	Δ Crop yield (% from mineral fertilization) [1]	-138	0	-71	52	-	-
Soil erosion	ΔSoil loss (%)[1]	-	-	-5	-36	-	-
	Δ Soil structural or aggregate stability (%)	29	41	0	63	-	-
Soil moisture content	Δ WHC (%)	0	50	-	-	-	-
	Δ PAW (%)	0	34	-	-	-	-
Soil workability	Δ Soil bulk density (%)[1]	-2.5	-21	-0.7	-23		-20
Soil biological properties and biodiversity[2]	Δ Microbial diversity (%)[1]	-	-	-	-	-2	4
	Δ Microbial biomass (%)	22	116	10	242	3.2	100
	Δ Microbial activity (%)	0	344	-	264	0	43
Crop nutritional quality	Crop nutritional quality (-)	nad	nad	-	-	-	-

Δ, change in the indicator; WHC, water holding capacity; PAW, plant available water; ns, no significant differences; nad, no average data because of complexity of available dataset; “-“, no reported benefits.

[1] Negative value indicates a decrease in the indicator.

[2] The ranges of benefit for three of the more used indicators are presented.

As shown in Table 1, losses of soil could be decreased by 5 to 36% with the application of compost, depending on the time horizon considered. A more precise quantification is possible for specific local conditions taking into account for instance climate, application rate, and type of soil. Avoided soil losses can be modelled within traditional LCA impact categories; here the consequential modelling should identify the agricultural production affected by the losses of arable land. Assuming a constrained agricultural production at a system level, the modelling is then done similarly to "crop yield", meaning that the consequences of intensification and/or land expansion are included in the assessment. Another alternative is to consider soil as a resource, thus either including loss of soil in the inventory as 'resource depletion' (Cowell and Clift 2000) (Figure1).

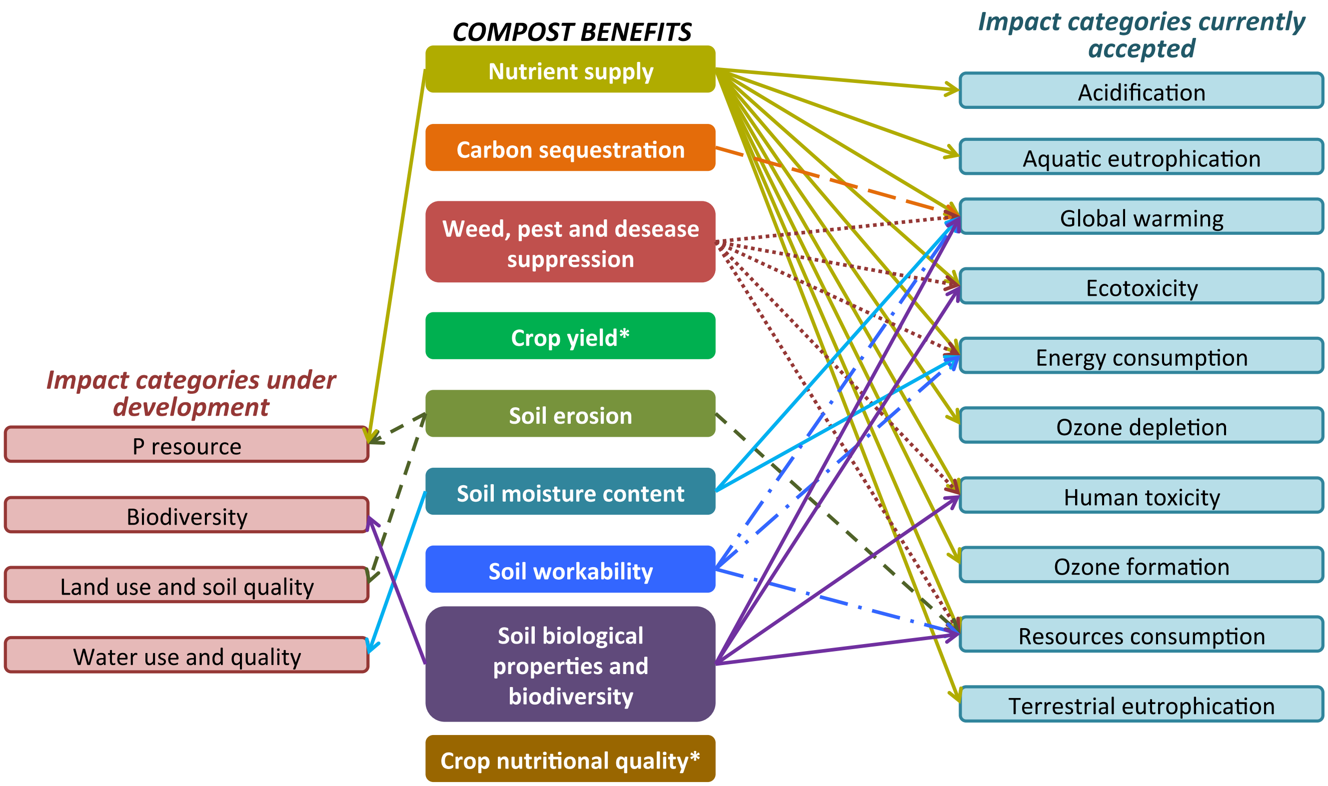

Figure 1. Midpoint LCA impact categories involved in the evaluation of the potential benefits of biowaste compost use-on-land.

* Could be considered as improvement of the function of agricultural processes and included in the functional unit definition.

Compost application might raise the capacity of soil to retain rainfall and irrigation water (green water, i.e. rainwater stored in the soil as soil moisture) allowing the reduction of irrigation water consumption (blue water, i.e. water from surface and groundwater resources). This may result in two distinguished consequences: on one hand, blue water is saved; on the other hand, because more green water is available, crop yield could increase in those areas where irrigation water is scarce. However, there is not a direct relation between the retention capacity and the amount of water saved, as it depends on the water demands of the plants, management practices, and previous moisture content of soils. The theoretical saving of irrigation water could be calculated for a particular case study if all these data are available (ROU 2007). Environmental burdens from irrigation water supply are linked to water extraction, transport, and distribution in the field (electricity, pumps, pipes, etc.), and are found in several inventories, also at a regional level. Potential impacts from these processes are typically those related to energy supply and consumption, thus Global Warming, Acidification, and Eutrophication categories (Figure 1). There is indeed a growing consensus on the fact that water use and consumption should be included in LCA assessing, apart from volumetric amounts, its environmental effects on the three Areas of Protection (Human Health, Natural Environment and Resources) (Núñez et al. 2013).

Improved soil workability can potentially decrease energy requirements for agricultural operations (Favoino and Hogg 2008; ROU 2007). Soil ploughing typically involves large consumption of energy and fuel. In an 8-year long study, McLaughlin et al. (2002) reported that, under a 100 t ha^{-1} application of stockpiled and rotted manure on a corn field, the plough draft was reduced 27–38%, resulting in 13–18% reduction of fuel consumption, due to the improvement in soil quality related to the increase in SOM. No other studies were found linking compost application and fuel consumption for agricultural operations, meaning that more comprehensive data are needed to be able to relate, for example, fuel consumption with soil bulk density. From an LCI point of view, reduced fuel consumptions can be credited to the system as avoided use of diesel. Avoided diesel consumption mainly affects Global Warming impact category, while avoided emissions of nitrogen oxide could also have an influence on Acidification and Eutrophication (Figure 1).

Changes in soil biodiversity after compost addition might influence either positively or negatively the services “delivered” by the ecosystem (i.e. nutrient cycling regulation of soil water and pest incidence), with consequences in terms of impacts associated to the substitution or compensation of those ecosystem services. In LCA terms, alterations in the system service in connection to biodiversity changes could be modelled within the traditional categories if those changes could be quantified in the inventory (Figure 1). If, for example, increased biodiversity can be directly related to increased nutrient cycling and lower need for fertilization, then the benefits from increased biodiversity could be modelled in terms of reduced production of fertilizers. However, data linking compost use, biodiversity and ecosystem services are non-existing, apart from a first attempt of establishing a preliminary relation by Nemecek et al. (2011). In addition, general figures cannot be established in all cases, as the effects of land management practices are highly variable depending on regional and scale dependent factors (Bengtsson et al. 2005). An alternative approach is to consider biodiversity and ecosystem services as independent endpoint categories when assessing the environmental impacts of land management alternatives (Zhang et al. 2010). Some recent initiatives have established baseline diversity indices for different soil organisms and under different soil uses that can be used as a reference to evaluate the impacts of compost on soil biodiversity (Cluzeau et al. 2012).

Different nutrient contents in food products resulting from compost application can have a repercussion on the LCA modelling depending on how the functional unit is defined. When the functional unit includes qualitative aspects (e.g. nutritional and/or economic value), increased nutritional level of a food product may have as a consequence that lower amounts of that specific food product are needed. In general terms, including qualitative aspects in the functional unit, would have an effect on the agricultural production (Martínez-Blanco et al. 2011), which could be modelled similarly to changes in crop yield.

4. Discussion

Regarding the environmental assessment – including quantification and characterization – of the benefits of compost application to soils, four different scenarios were identified: (i) The positive effects of compost application are proved, effects are quantifiable, and there are tools for their consideration with LCA. This includes nutrient supply and carbon sequestration, which are (and should be) included in LCA studies. (ii) The benefits are proved, but their magnitude is too variable as a consequence of the synergetic effect of many factors. Thus, inventory data cannot be unambiguously quantified. Impact categories and characterization factors exist for most of the benefits. (iii) The benefits are proved and quantifiable. However, corresponding characterization factors and/or impact categories are non-existing. (iv) Benefits are not fully proved and thus their inclusion in the modelling is not yet feasible.

Two out of the nine potential benefits initially proposed, are proved and the quantification and assessment are possible, while different research efforts are required for the rest of the effects for a full assessment both regarding improved modelling and characterization. Modeling and quantification issues are related to the fact that LCA models are typically linear steady-state models of physical flows (Guinée et al. 2002) whereas fluxes of nutrients and pollutants after compost application to soil are not linear in most of the cases. This also applies, for instance, to repeated applications of compost: LCA studies typically look at the effects of a single application over 100 years, while the cumulative effects on several applications may not be linear with the amount of compost added. Also, LCA models assume that impacts depend on the compost characteristics while they rarely include environmental parameters as determining factors, which highlights the necessity of coupling LCA and agronomic models to gain a more precise picture. Another methodological issue is derived from the fact that many of the benefits discussed in this paper might be interrelated and therefore their contribution to a specific impact category might be overlapping. This should be clearly identified in order to avoid the overestimation of the benefits.

Finally, for LCA of the agricultural sector, the functional unit is typically defined per area used or product yield. As different functional units can lead to different results for the same product system (Martínez-Blanco et al. 2011), there may be a need for a qualitatively more precise definition when dealing with compost application, especially in those cases where the product quality is affected. Better definitions could, for example, include the economic value or the nutritional content of a product (Schau and Fet, 2008). A more accurate definition dealing with nutritional differences may include a combination of quality (nutritional quality) and quantity (yield). This was for example done in Charles et al. (1998) and Audsley et al. (2003), where the functional unit was defined as "1 equivalent ton grain with 12–13% protein". This involved the use of marginal productions to adjust the overall output of the system under assessment. Finally, the choice of the time horizon of the LCA should be harmonized. The studies reviewed showed in fact that such choice is in many cases very important, as both the foreground and background effects of compost application vary largely depending on the time frame.

In addition to improved quantification and modelling of compost application, the development of new impact categories or modifications of the current ones in the future will allow for a more comprehensive assessment of compost benefits. These should deal with depletion of P resources, biodiversity, loss of arable soil, and consumption of water. Depletion of P as a resource is currently modelled similarly to other natural resources. A revision of the characterization factors is thus needed for the assessment of non-replaceable non-renewable resources such as P. In this respect, the ReCiPe model adds value to resources; it is based on the geological distribution of mineral and fossil resources, and assesses how the use of these resources causes marginal changes in the efforts to extract future ones (Goedkoop et al., 2009).

Although there is no consensus yet on which indicators use in the assessment of land use impacts, there is a common agreement that land use is one of the main drivers of biodiversity loss, and that must be assessed taking into account different taxonomic groups and a spatially explicit approach (De Baan 2013).

Soil loss involves the loss of cultivable land but also the loss of soil organic carbon (SOC), plant nutrients, as well as the associated plant, animal and microbial biodiversity (Cowell and Clift 2000). Loss of soil can thus be included in some of the abovementioned impact categories. However, for a more comprehensive assessment the loss of soil mass could be considered as the loss of a resource and included in the inventory as 'resource depletion' (Núñez et al. 2013). In alternative, soil erosion can be included within the impact category Land Use, whose characterization factors are based on soil quality indicators such as SOM, structure, heavy metals, biodiversity, aesthetic value, etc. (Brentrup 2004; Mattsson et al. 2000).

Depletion of water resources is gradually gaining importance, particularly in certain geographical regions. There is currently only a preliminary scientific consensus about the parameters to consider and the methodology to follow (Núñez et al. 2013). Methodological issues concerning impact assessment methods include the types of water use accounted for, the inclusion of local water scarcity conditions, and the differentiation between watercourses and quality aspects (Berger and Finkbeiner 2010).

5. Conclusion

Most potential environmental benefits of compost are so far not included in LCA studies because of scarcity of data, high variability in the observed effects, or lack of appropriate impact assessment methods. For two of the nine benefits – nutrient supply and carbon sequestration – the literature review showed that both quantification and impact assessment of the effects could be performed, meaning that these two benefits should be regularly included in LCA studies. For four of the nine benefits – increase in crop yield, soil workability, crop nutritional quality, and enhancement of soil biological properties and biodiversity –, quantitative figures could not be provided, either because of complete lack of data or because the effects are both very variable and too depending on specific local conditions. For "soil erosion" and "soil water content" effects could be quantitatively addressed, but available impact assessment methodologies were considered unsuitable to comprehensively evaluate the implication of compost application with regards to these two benefits. Finally, based on the available literature, "suppressive effects of compost on weed, pests, and diseases" could not be generally proved. Efforts at different levels are needed in order to comprehensively evaluate the benefits of compost use, such as the collection of more empirical data to accurately determine the magnitude of some of the effects. Long-term studies are particularly scarce. Further, the comprehensive assessment of compost benefits would also need further improvement of the modeling for the quantification of the benefits, as well as a better understanding of how the local environmental conditions would influence the effects of compost, through the use of agronomic models. Additional impact categories dealing with phosphorus resources, biodiversity, soil losses, and water depletion, may be required.

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Implications of alternative greenhouse gas metrics for life cycle assessments of livestock food products

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ABSTRACT

Life cycle assessments of livestock food products incorporate emissions of different greenhouse gases (GHGs) with very different physical properties. These are commonly aggregated into a single unit, called "CO_2-equivalent", using the 100-year Global Warming Potential. However, an increasing literature emphasizes that many alternative metrics can be justified to compare emissions of different GHGs and would result in a substantially different weighting for methane relative to nitrous oxide and carbon dioxide. We demonstrate that alternative GHG metric choices can materially alter the difference in emissions associated with organic, high-input and conventional dairy farms; and that metrics can substantially shift the differences in GHG footprints of ruminant and non-ruminant meat products. We conclude that the characterization of certain products and farm systems as more or less climate-friendly reflects, in part, implicit choices and value judgments made in the selection of GHG metrics.

Keywords: greenhouse gas metric, Global Warming Potential, livestock, dairy production, organic systems

1. Introduction

Life cycle assessment (LCA) of food products covers various environmental impact categories, including effects on climate. This category needs to incorporate various greenhouse gases (GHGs) with very different physical properties such as differing lifetimes in the atmosphere and contributions to the warming of Earth's atmosphere. To enable comparisons between different food products and production systems, emissions of individual gases are aggregated into a single characterization factor (called 'Global Warming Potential'), using the unit of "CO_2-equivalent" emissions. The 100-year Global Warming Potential (GWP_{100}) is the most commonly used metric to convert emissions of individual gases into CO_2-equivalent emissions; it is employed in reporting and accounting under the United Nations Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol and in the great majority of LCAs related to climate change.

However, an increasing literature emphasizes that the GWP_{100} is only one of many alternative metrics that can be devised to compare emissions of different GHGs (e.g. Bowerman et al. 2013; Manning and Reisinger 2011; Shine 2009; Smith et al. 2012; Tanaka et al. 2010). The most appropriate choice of metric depends crucially on policy objectives (IPCC 2009; Myhre et al. 2013), and the current use of GWP_{100} for reporting emissions under the UNFCCC is not necessarily consistent with the ultimate objectives of the Convention (Tol et al. 2012). The most recent assessment by the Intergovernmental Panel on Climate Change (IPCC 2013) does not recommend the GWP_{100} as metric of choice, but presents a range of alternative options (Myhre et al. 2013); the metric for a future global climate agreement is being actively debated in the UNFCCC (e.g. UNFCCC 2012).

Alternative climate metrics, reflecting different policy goals and value judgements, would result in different weightings for short-lived GHGs, most notably methane (CH_4). This has potentially significant implications for the LCA of food production, given that CH_4 and nitrous oxide (N_2O) constitute large fractions of the total emissions from livestock-based agriculture. As a result, well-publicised differences in life cycle GHG emissions for different agricultural products and processes could change significantly if a metric other than the GWP_{100} were chosen, since different farm systems typically exhibit different fractions of emissions of CH_4, N_2O and CO_2. Consequently, the relative 'climate-friendliness' of particular farm systems or foods could depend strongly on the choice of emission metric. The importance of alternative climate metric choices for LCA has been discussed in some publications (e.g. Peters et al. 2011), but few studies have explored implications for LCA studies of food production and strategies to reduce agricultural emissions (for one example, see Reisinger and Ledgard 2013).

To advance this discussion, we select in this study a range of published climate metrics and apply them to a range of published food LCA studies. We demonstrate that the choice of alternative metrics indeed has a material impact on the relative climate-friendliness of different farm systems (organic vs conventional and high-input dairy farms) and of different meat products (particularly meat from ruminants or monogastric animals). We demonstrate that the choice of climate metric is at least as important as other methodological and parameter choices that are, in contrast to climate metrics, discussed widely in the LCA literature.

Section 2 presents the metrics used in this study. Section 3 applies those metrics to different dairy production systems and also explores their implications for the climate footprint of various meat products. Section 4 discusses the results and considers their implications for mitigation strategies, product labelling and possible consumer choices geared towards climate-friendly food production, and considers results in the context of food miles. Section 5 concludes, including a short outlook to additional work.

2. Selection of climate metrics and their use in food LCA studies

A recent summary by Tanaka et al. (2013) demonstrates that even if there is agreement to a long-term global climate goal, a large range of choices remains for climate metrics. Scientifically justifiable weighting values for CH_4, relative to CO_2, range from about 4 to more than 70. Furthermore, advances in scientific understanding and changes in atmospheric background concentrations of GHGs result in the best estimates of metric values themselves changing over time (Joos et al. 2013; Myhre et al. 2013; Reisinger et al. 2011). The purpose of this study is not to present the most up-to-date estimates of climate metrics or a comprehensive overview, but rather to test the sensitivity of key conclusions of some food LCA studies to metric choices other than the default GWP_{100}. To this end, we select two alternative metrics and time horizons that have received considerable attention and scrutiny in the scientific literature in addition to GWP_{100}, namely the GWP with a shorter time-horizon of only 20 years (GWP_{20}), and the Global Temperature change Potential with a time horizon of 100 years (GTP_{100}). Scientific justifications and analysis of these metrics, and the calculation of specific metric values, can be found elsewhere (see e.g. Fuglestvedt et al. 2003; IPCC 2009; Shine et al. 2005; Tanaka et al. 2010). Table 1 presents the values for those metrics for CO_2, CH_4 and N_2O as used in the reminder of this study.

Table 1. Metric values for GWP_{100}, GWP_{20}, and GTP_{100} as used in this study. Values are based on Forster et al. (2007) for GWP_{100} and GWP_{20}, and on Fuglestvedt et al. (2010) for GTP_{100}.

Gas	GWP_{100}	GWP_{20}	GTP_{100}
CO_2	1	1	1
CH_4	25	72	4
N_2O	298	289	265

We emphasize that the metrics and values chosen in this study are intended to serve a sensitivity analysis of LCA results; they are not intended to represent the most up-to-date or appropriate choices. Some metric choices, such as GWP_{500}, are not included because the numerical weighting for CH_4 metric would be quite similar to that using GTP_{100} even though the scientific rationale leading to this value is different. We also note that the current best estimate for the 100-year GWP of CH_4 is 28 (Myhre et al. 2013), updated from the previous estimate by the IPCC of 25 in 2007 (Forster et al. 2007) and 21 as used to date under the UNFCCC and the first commitment period of the Kyoto Protocol based on IPCC (1996). In this study, we use the values for GWP_{100} as published in Forster et al. (2007) to facilitate comparability with other LCA studies that have used these values.

Many published LCA studies using GWP do not provide emissions of individual gases in their native units, but the use of GWP_{100} is often ‘hard-wired’ into results. This can make it difficult to separate out the effect of climate metric from the overall conclusions. In this analysis, we use a small number of published studies where we had access to the raw calculation of emissions of individual gases, and hence the implications of alternative metrics can be explored. Similar to the choice of metrics, the selection of studies or food production systems in this analysis is not intended to be comprehensive or representative of the selected issues, but merely to demonstrate the potential sensitivity of LCA-based GHG studies to the choice of climate metrics – but equally to show where conclusions might be robust regardless of the climate metric used. Further work will be required to test the extent to which the conclusions derived from these limited examples can be generalised.

3. Results

We explore the implications of climate metrics in two specific applications: (1) the relative climate-friendliness of organic, conventional and high-input dairy production systems, using a review across different countries and two detailed analyses from New Zealand; and (2) the carbon footprint of different meat products from ruminant and monogastric animals, using a previously published European study as input.

3.1. Carbon footprint of milk from organic, conventional and high-input dairy production

We explored the implications of alternative metrics for the carbon footprints of conventional, organic and other environmentally-friendly milk production systems, based on a review across different countries (de Boer 2003). The original study showed that organic systems had a smaller carbon footprint per kg of fat and protein corrected kg of milk in Sweden, but similar or greater footprint in the Netherlands and Germany, reflecting different production systems as well as potentially different parameter assumptions. More recent studies reported similarly mixed conclusions (Cederberg and Flysjö 2004; Kristensen et al. 2011; Thomassen et al. 2008), with results strongly dependent on the methodology used for land-use change (Flysjö et al. 2012). If GTP_{100} were used in the review by de Boer (2003) to calculate carbon footprints, organic systems would in all three countries show a lower climate footprint than conventional systems, but a consistently greater footprint if GWP_{20} were used. Interestingly, other environmentally friendly/extensive farm systems reviewed in de Boer (2003) show a lower climate footprint for all three climate metrics. This suggests that the choice of metric might be at least as important as country-specific differences in production systems and parameter choices relating to land-use change in determining whether organic systems are more or less climate-friendly than conventional ones.

To further test this conclusion, we examined the sensitivity of the LCA of cradle-to-farm-gate GHG emissions associated with milk production in different farm systems. The first evaluation covers the carbon footprint at the farm gate of milk produced in typical organic, high-input and average (conventional) dairy farms in New Zealand. The organic and high-input (about 25-40% of feed brought-in from off-farm) farm systems were based on survey data from representative groups of 10-20 dairy farms in the North Island of New Zealand (Ledgard et al. 2009). The average dairy farm was based on survey data (animal population, productivity and farm inputs from the DairyBase database; DairyNZ 2014) from over 200 farms from the various dairying regions throughout New Zealand for 2010/2011, weighted for relative milk production (as described in Flysjö et al. 2011). Total GHG emissions were allocated between milk and meat using the IDF (2010) methodology.

The original study showed that organic farms had a lower carbon footprint than conventional farms and significantly lower footprint than high-input farms, if GWP_{100} is used to calculate carbon footprints. Most of this difference comes from off-farm GHG emissions (from fertilizer, brought-in feed, and off-farm grazing), which are minor for organic farms but constitute about 10% for conventional and more than 20% for high-input farms.

However, these conclusions are sensitive to the choice of metric, as summarized in Table 2. Organic farms tend to have a higher percentage of CH_4 emissions per kg of milk solids (fat plus protein) due to the less intensive feeding regime, but a lower percentage of N_2O emissions due to nil use of nitrogen fertilizer (relying on clover N_2-fixation), negligible use of brought-in feed, including associated CO_2 emissions from fossil fuel consumption. Since alternative metrics change the relative weight given to the short-lived gas CH_4 relative to the longer lived gases N_2O and CO_2, the net emissions from different farm systems depend significantly on the choice of metric. Organic farms have 13% lower emissions than conventional farms under GWP_{100}, but 30% lower emissions under GTP_{100} and only 5% lower emissions under GWP_{20}. This is because the latter metric gives much greater weight to the higher CH_4 emissions per kg of product on organic farms. The picture is reversed for high-input dairy farms, which have 11% higher emissions than conventional farms under GTP_{100} but only 2% higher emissions under GWP_{20}, because GTP_{100} gives the greatest relative weight to N_2O emissions which dominate on high-input farms due to the high use of nitrogen fertilizers.

A related shift occurs in the balance between on-farm and off-farm emissions. Off-farm emissions are always a small percentage of total LCA emissions for organic farms, but can range from 41% under GTP_{100} to 13% under GWP_{20} for the high-input farms, and from 20% under GTP_{100} to 6% under GWP_{20} for conventional farms. These different fractions arise because most of the off-farm emissions arise in the form of long-lived gases N_2O and CO_2, which are given much greater weight if GTP_{100} is used as a metric than under GWP_{20}.

Based on this analysis, we conclude that for the specific parameters and input factors used in this study, the milk of organic farms has a lower carbon footprint, and high-input farms a higher carbon footprint than the average conventional dairy farm in New Zealand regardless of the climate metric chosen, but the amount of difference depends significantly on the metric. If only the on-farm component of emissions were considered, different climate metrics would suggest that the climate-benefit of organic farming could range from 18% lower emissions than conventional farms (using GTP_{100}) to almost no difference (only 1% lower emissions using GWP_{20}).

Table 2. On-farm, off-farm and whole-farm system LCA GHG emissions from milk from conventional, organic and high-input New Zealand dairy farms, in kg CO_2-eq per kg of milk solids produced over the cradle-to-farm-gate stages, for GWP_{100}, GTP_{100} and GWP_{20}. Columns in italics show the emissions from organic and high-input systems relative to conventional average of farms.

Scope	**GWP_{100}**			*Org. /*	*High/*	**GTP_{100}**			*Org. /*	*High/*	**GWP_{20}**			*Org. /*	*High/*
	Conv.	Org.	High	*Conv.*	*Conv.*	Conv.	Org.	High	*Conv.*	*Conv.*	Conv.	Org.	High	*Conv.*	*Conv.*
whole farm	10.5	9.1	11.0	*-13%*	*4%*	4.8	3.3	5.3	*-30%*	*11%*	22.7	21.5	23.1	*-5%*	*2%*
on-farm	9.5	8.9	8.5	*-6%*	*-10%*	3.8	3.1	3.1	*-18%*	*-18%*	21.4	21.3	20.1	*-1%*	*-6%*
off-farm	1.1	0.2	2.4	*-80%*	*129%*	1.0	0.2	2.2	*-79%*	*128%*	1.3	0.2	3.0	*-83%*	*133%*
Percentage off-farm	10%	2%	22%			20%	6%	41%			6%	1%	13%		

These conclusions are supported by another dataset, which looked only at the on-farm carbon footprint of a specific set of 23 different conventional and organic dairy farms in New Zealand (Barber 2010). In this study, using GWP_{100}, average on-farm emissions per kg milk solids from organic dairy farms were found to be almost identical to those from conventional farms, but with a considerable variability between farms within each farm system (Figure 1). However, the average organic farm would show 8% lower emissions than the average conventional farm if GTP_{100} were used as the metric, but 4% higher emissions if GWP_{20} were used. These results are consistent with organic farms typically higher CH_4 emissions per kg of milk solids owing to lower milk yield per cow, but lower N_2O and CO_2 emissions mainly due to the lower nitrogen fertilizer use. This higher fraction of CH_4 emissions in organic systems is weighted more strongly by GWP_{20} but less if GTP_{100} is used.

Collectively, these results show that the choice of climate metric could have a significant influence on whether organic farming is considered more or less 'climate-friendly' than average conventional practices. However, these conclusions apply only to the 'average' farm in both systems; regardless of metric, within this sample there is always a conventional farm with a lower GHG footprint than the most efficient organic farm (Barber 2010).

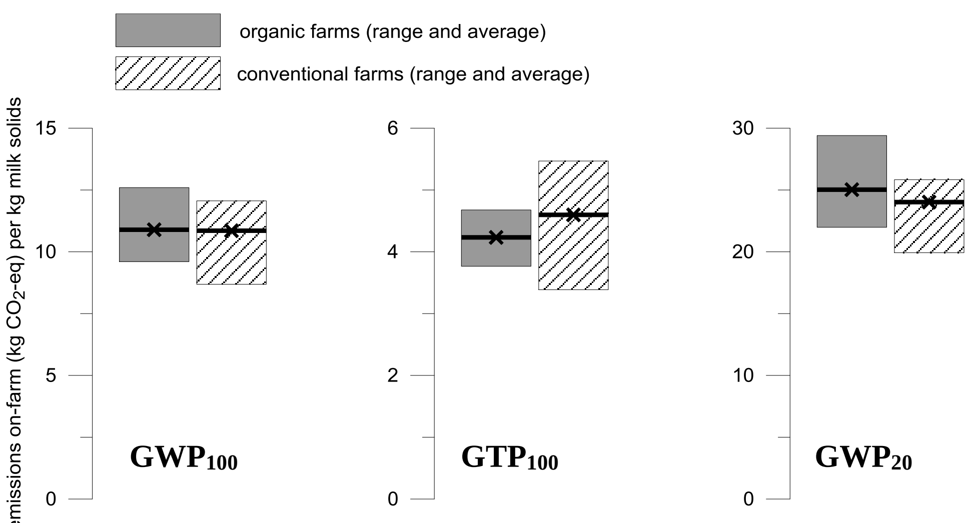

Figure 1. Average, maximum and minimum on-farm GHG emissions from 23 individual organic and conventional dairy farms, for GWP_{100} (left panel), GTP_{100} (center panel), and GWP_{20} (right panel). Note Y-axis has been scaled for each metric to reflect different absolute CO_2-eq emissions for different metrics. The black horizontal bar indicates the average across the surveyed farms for each farm system.

3.2. Climate-footprint of different meat products

LCA can be used to inform policymakers, industry and consumers about whether particular products are, in general, more or less ‘climate-friendly’ than alternative products. A consistent conclusion across many LCA studies is that meat from ruminants (mainly beef, sheep and goat meat) has significantly higher GHG emissions per kg of product at the farm gate than meat from monogastric animals (mainly pork and poultry). This implies that diet switching could be an important part of global strategies to reduce GHG emissions from agriculture (Cederberg et al. 2013; de Boer et al. 2011; Hedenus et al. 2014; Ripple et al. 2014).

We tested the robustness of this conclusion against the choice of different climate metrics, using a comprehensive LCA of livestock production in the European Union (Weiss and Leip 2012). The original study confirmed that farm-gate GHG emissions per kg of meat from ruminants (cattle, sheep and goats) were significantly higher on average across Europe than emissions per kg of monogastric animals (pork and poultry), but the study also showed that emissions per kg of product varied significantly across the 27 member countries. Assumptions around land-use and land-use change emissions for different products and countries were found to have a major impact on the relative ranking of countries in terms of their GHG efficiencies, even though the overall result of ruminant meat being more emissions-intensive on average than monogastric meat was not affected.

We recalculated the average emissions per kg of beef, sheep and goat meat, pork and poultry for the average across the 27 European member states included in the study, for GWP_{100}, GTP_{100} and GWP_{20}. Figure 2 shows CO_2-eq emissions per kg of sheep/goat meat, pork or poultry *relative to* the CO_2-eq emissions per kg of beef, for each of the three different metrics and across the different land-use and land-use change scenarios used in the study. The Figure also shows results if emissions from land-use and land-use change are excluded. The relative ranking of the emissions intensity of different meat products is robust across the three different metrics, with emissions per kg sheep and goat meat comparable to emissions per kg of beef, but pork having consistently lower emissions and poultry the lowest. However, the choice of metric has a large impact on how much lower the GHG emissions from monogastric meat production are compared to those from ruminant meat production. If GWP_{20} is used as climate metric, CO_2-eq emissions per kg of pork are only 20% of the emissions per kg of beef, but almost 50% of beef emissions if GTP_{100} is used. The differences due to alternative metrics choices are comparable to the difference between direct emissions and those from additional land use and land-use change, and considerably larger than the range of results under different land-use and land-use change scenarios.

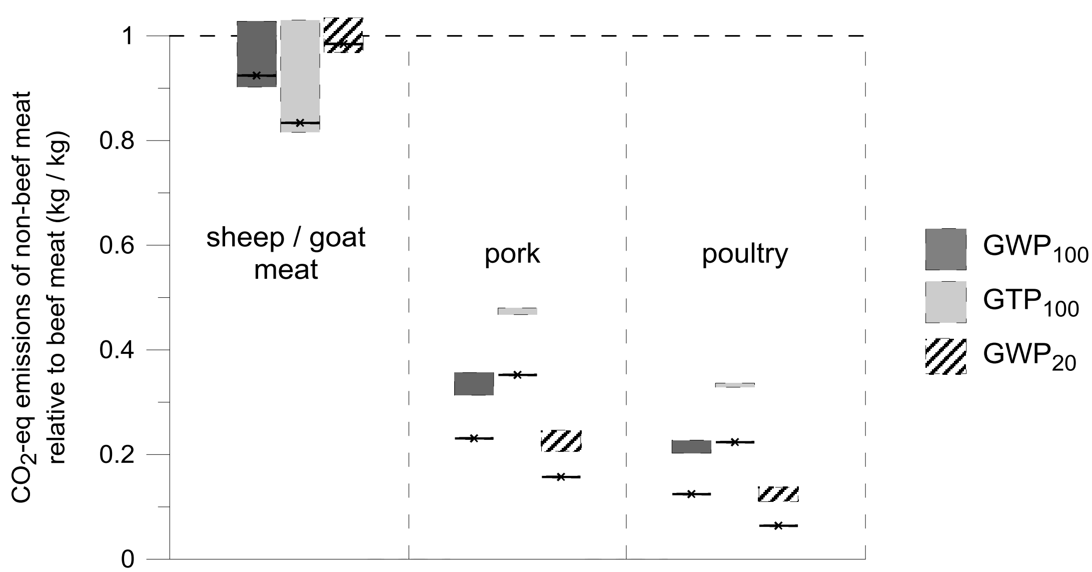

Figure 2. GHG emissions per kg of sheep and goat meat, pork and poultry relative to emissions per kg of beef. Emissions were calculated in CO_2-eq using three different climate metrics, based on data from Weiss and Leip (2012) for the average of 27 EU countries. The vertical bars show, for each metric, the range of results based on different land-use and land-use change scenarios. The black horizontal lines show the emissions per kg of meat relative to beef if contributions from land-use and land-use change are excluded from the calculations.

The emissions of sheep/goat meat relative to beef are much less influenced by alternative metrics and depend more on different land use and land-use change scenarios. These results are plausible, as a significant fraction of emissions for both beef and sheep/goat meat production arises from CH_4 from enteric fermentation.

We also tested the influence of metrics on the ranking of countries within the European Union based on their emissions intensity of meat production. As demonstrated by Weiss and Leip (2012), countries differ significantly in how CH_4 from enteric fermentation, N_2O and CO_2 from both energy use and land use/land-use change contribute to overall GHG emissions of ruminant meat production in individual countries. The use of different metrics, which apply different weights to CH_4 compared to the longer lived gases N_2O and CO_2, can materially change this ranking, as demonstrated for beef in Figure 3 (top panels). For GTP_{100}, the countries with the highest emissions per kg of beef remain largely unchanged, since their ranking is dominated by high CO_2 emissions from land-use/land-use change, which are given even more emphasis under GTP_{100}. Differences arise mainly in the ordering of the lowest and some middle emissions producers, although Austria remains as the country with the lowest emissions. Use of GWP_{20} as metric would result in more significant re-ordering of countries, as some countries with high CH_4 emissions become high emitters per kg of product even though they have low CO_2 emissions from land use/land-use change. At the same time, some countries with low CH_4 emissions become low emitters overall, even if they show more significant land use/land-use change emissions; the Netherlands replaces Austria as the country with the lowest emissions per kg of product. The GWP_{20} values also showed less variation across countries (approximately 2-fold) than for GWP_{100} (3-fold) or GTP_{100} (5-fold), because direct emissions from enteric fermentation (which are emphasized if GWP_{20} is used) are more similar between countries than the additional emissions related to land-use/land-use change.

Such changes in rankings of countries are much less pronounced for emissions per kg of pork (see Figure 3, bottom panels). This is as expected, given that CH_4 constitutes a much smaller fraction of emissions and alternative metrics have a much smaller influence on the weighting of N_2O relative to CO_2.

Figure 3 also demonstrates that the choice of alternative climate metrics influences whether and to what extent the country of origin matters in determining the relative climate-friendliness of pork compared to beef. If GWP_{100} is used, pork mostly has a lower carbon footprint than beef; only two countries with the highest emissions intensities produce pork with a similar carbon footprint (at 15-20 kg CO_2-eq per kg of meat) as the six countries with the lowest emissions intensities for beef. If GTP_{100} is used, the overlap is much stronger: pork from the two countries with the highest emissions intensities now has a carbon footprint (at 14-20 kg CO_2-eq per kg of meat) that is similar to or more than the footprint of beef from more than half of the 27 EU member countries. By contrast, if GWP_{20} is used, pork always has a lower carbon footprint than beef for any country.

4. Discussion

The results from our analysis demonstrate that the choice of alternative climate metrics has a material influence on the question of whether organic dairy farm systems have a lower carbon footprint than conventional or high-input systems, and substantially alters the magnitude of the difference between the carbon footprint of meat from ruminants compared to meat from monogastric animals. Given differences in carbon footprints between different countries for the same meat product, beef from some countries could have a lower carbon footprint than pork from some other countries with relatively high emissions intensity, with the outcome depending strongly on the climate metric chosen to account for GHG emissions. Carbon footprinting is intended to support mitigation strategies, product labelling and possible consumer choices geared towards climate-friendly food production, including choices regarding diet and country of origin. Our results demonstrate that the choice of climate metric has a material effect of whether such choices are robust, but the metric is often not communicated, let alone justified. At the least, a sensitivity analysis illustrating effects of several metrics is desirable. GWP_{100} is but one of many choices that can be justified in some respects using scientific arguments, but its appropriateness for any application depends on the goals and decisions that any specific life cycle assessment intends to support.

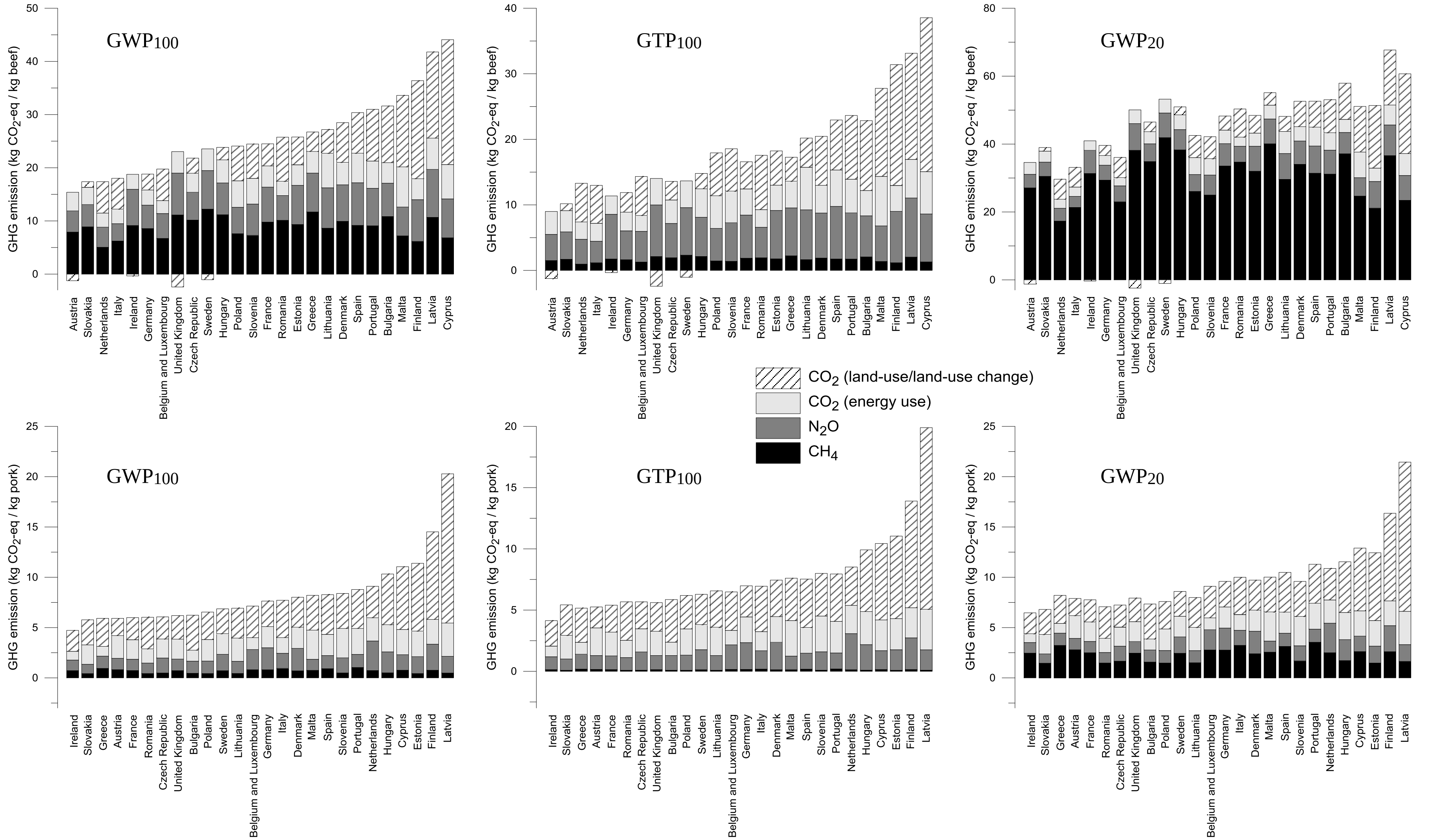

Figure 3. Impact of alternative climate metrics on the ranking of European member countries (EU-27) in terms of their emissions intensity of meat production (kg CO_2-eq / kg meat) for different climate metrics. Raw data are from Weiss and Leip (2012), using scenario II for emissions from land-use and land-use change. The top panels show results for beef, bottom panels for pork. Left panels show countries ranked based on GWP_{100} (using metric values of 21 for CH_4 and 310 for N_2O as in the original study), while middle and right panels show the same order of countries, but with their emissions intensities calculated using GTP_{100} and GWP_{20}, respectively.

Our results also suggest that the importance of food miles, i.e. emissions due to the transport of food to distant markets, can depend strongly on the climate metric. As an example, a study by Flysjö et al. (2011) indicated that the carbon footprint of milk at the farm gate was about a 15% lower on average in New Zealand than in Sweden if GWP_{100} is used. Once emissions arising from the processing of milk and its transport to Europe are taken into account (Hutchings and Ledgard 2009), the total carbon footprint of New Zealand milk sold in Sweden would be some 5% higher than for milk produced and sold in Sweden. However, if GTP_{100} were used, milk production in New Zealand would have 31% lower emissions at the farm gate compared to Swedish milk, given that New Zealand milk has a higher fraction of emissions in the form of enteric fermentation, which is valued less in the GTP_{100} metric (Flysjö et al. 2011). As a result, if GTP_{100} is used, the overall carbon footprint including emissions related to milk processing and transport from New Zealand to Sweden would be almost identical between the two countries. This cursory comparison, and our analysis of the data from Weiss and Leip (2012) within the European Union, indicate that at least for some examples, the choice of metric could have a significant influence on how significant food miles are in comparisons of the overall carbon footprint of products produced in different countries and traded internationally to reach distant markets.

5. Conclusion

Life cycle assessment of the climate impact of different food products and different farm systems requires a metric to convert the emission of different GHGs into a common unit. Even though GWP_{100} is by far the most commonly used metric, many others can be scientifically justified and could be more appropriate, depending on the specific application and decisions that the LCA seeks to support.

We used three alternative metrics, which differ significantly in the weighting for emissions of CH_4 relative to CO_2, to demonstrate that the choice of climate metric materially influences the question whether organic dairy farm systems have a lower carbon footprint than conventional or high-input systems. Depending on the specific study, the choice of metric can flip the result entirely, such that organic systems have lower footprints if GTP_{100} is used but higher footprints if GWP_{20} is used. The carbon footprint of beef from ruminants, based on one comprehensive European study, is higher on average than meat from monogastric animals regardless of the metric choice, but the difference depends substantially on the choice of metric. Given country-country variations, the metric choice could result in beef from some countries having a lower carbon footprint than pork from other countries within the European Union depending on the climate metric chosen to account for GHG emissions.

The choice of metric thus can substantially influence the relative climate-friendliness of different food products and farm systems. The influence is comparable in magnitude to whether and how emission related to land-use and land-use change and emissions related to international transport of food products are accounted for.

We have used only a limited number of readily available studies to test the sensitivity of their results to the choice of climate metric. It would be highly desirable to test our results against a wider range of studies with different assumptions. Given our results, the choice of climate metric deserves much greater attention in LCA studies and transparency when key conclusions about GHG footprints from different farm systems or food products are communicated. To support such transparency, LCA of GHG emissions should strive to retain emissions of individual gases in their ‘native’ units to allow the sensitivity of results to be tested against different metric choices, and results to be updated as metric values change.

6. Acknowledgements

The authors would like to thank Adrian Leip for kindly providing the full dataset underpinning the study by Weiss and Leip (2012), which provides the basis for our analysis of the impact of climate metrics on the carbon footprints of different meat products within the European Union.

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Learnings from an FAO-led international process to develop LCA guidelines for small ruminants: A LEAP Partnership initiative

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ABSTRACT

Methodology and guidelines for quantifying greenhouse gas (GHG) emissions and fossil fuel demand from sheep and goat supply chains covering the system boundary of the cradle-to-primary-processing-gate were developed by an international technical group. Key learnings included the need to recognize and account for 1) the global diversity of small ruminant production and processing systems, 2) the potential for limited data availability, and 3) several main areas of methodology that have potential to change the final results. Two critical areas of Life Cycle Assessment (LCA) based methodology identified were the importance of determining feed requirements by animals and the methods for handling multi-functional processes including allocation. On-farm, biophysical allocation based on relative feed requirements was recommended for allocation between animal species sharing the same feed sources and for allocation between meat, milk and/or fiber. However, sensitivity analysis of different allocation methods should be carried out and results presented to illustrate the effects on results.

Keywords: greenhouse gases, goats, international, methodology, sheep

1. Introduction

Livestock have been identified as being major contributors to resource use and environmental impacts at a global scale (e.g. Gerber et al. 2013; Steinfeld et al. 2006). Small ruminants (e.g. sheep and goats) are a relatively small component of the total livestock sector but are of global importance because they cover a wide diversity of systems across an enormous range of geographical regions that provide a variety of products and functions. In 2011, sheep and goats produced more than 5 million tonnes of meat and 24 million tonnes of milk globally and production has increased by 1.7% and 1.3% per year, respectively, during the past 20 years (FAOSTAT, 2014). This increase was driven mainly by developing countries in Africa and Asia, although Oceania (mainly for meat) and Europe still contribute significantly to production. Production systems can vary from intensive systems, where animals are partially or predominantly housed, to extensive systems which rely on grazing and native forages, and transhumance systems that involve large flock movements. Products are not restricted to meat and milk; sheep are also valued for their wool (more than 2 million tonnes of greasy wool was produced globally in 2011), and goats for their mohair and cashmere. Small ruminants also play a crucial role in small-scale, rural and family-based production systems by sustaining livelihoods, contributing to food security and nutrition, providing a way to store and manage wealth and supporting ecosystem services.

Of the range of environmental impacts from livestock systems, climate change associated with greenhouse gas (GHG) emissions has received the most attention during the past decade. This has resulted in efforts to support decision-making of purchases by consumers by making them aware of GHG emissions linked to products they eat through environmental labelling of products (e.g. Tescos scheme in the United Kingdom and one of the indicators in the pending Grenelle scheme in France). Labelling focused on the carbon footprint (i.e. total GHG emissions expressed on a CO_2–equivalent basis) throughout all stages of production and provision of products to consumers has generally involved the use of a Life Cycle Assessment (LCA) approach.

Similarly, an array of environmental assessment methods have been set up in support of product-based environmental performance schemes for business-to-business communication (e.g. environmental product declaration schemes), and of environmental improvement reporting systems (e.g. incentives linked to GHG emission mitigation options). Furthermore, other LCA-based approaches are being examined by some governments to monitor the environmental footprint of the economy and set environmental policy priorities at the sector level accordingly (e.g. the life-cycle based indicators developed by the European Commission).

Given such proliferation, it is desirable that an internationally-acceptable common methodology is used so that products can be assessed on a similar basis. Common and robust methodology tailored to the specific nature of small ruminant supply chains will also enable various stakeholders to identify hotspots and opportunities to reduce environmental impacts. Some sectors have started to work together to agree on a methodology, which has typically been based on use of an LCA approach (e.g. for milk products by the dairy sector; IDF 2010). Similarly, initial work on a draft methodology for lamb meat was initiated by Beef+LambNZ and the International Meat Secretariat. Recently, the Food and Agriculture Organization of the United Nations (FAO) has initiated a broad process to develop harmonized international methodologies and guidelines to assess the environmental performance of livestock supply chains. This resulted in the Livestock Environmental Assessment and Performance (LEAP) Partnership, which is a multi-stakeholder initiative whose goal is to improve the environmental sustainability of the livestock sector through better metrics and data (LEAP 2014a). The LEAP Partnership comprises a large range of government, industry and civil society organizations. The three groups are represented at a Steering Committee, while the Guidelines are primarily developed by Technical Advisory Groups (TAGs) of international experts with experience in LCA, GHG emissions and livestock systems. One of such TAGs was established and has been active over the past year in developing guidelines for assessing GHG emissions and fossil fuel demand for the small ruminant sectors. This paper reports on aspects of methodology development of the small ruminant guidelines and key learnings from it. In March 2014, the draft guidelines for small ruminant supply chains (LEAP 2014b), poultry supply chains (LEAP 2014c) and animal feeds (LEAP 2014d) were released for public review by the FAO, with the intention of revising them and producing a final set of guidelines in late-2014.

2. Methodology

The first key role of the TAG was to define the scope of the methodology. It was decided to limit the small ruminants covered by the guidelines to goats and sheep due to their significance and data availability. However, the principles developed could be applied to other small ruminant species such as alpaca and deer. It was recognized that globally there are a very wide range of products produced from small ruminants with the major ones being meat, milk and fiber. It was decided to confine the system boundary covered by the guidelines to the cradle-to-primary-processing gate (Figure 1), while recognizing that environmental communication and comparisons of products must be based on the full supply chain and, therefore, the importance of future development to extend the guidance to the full life cycle. The primary processing stage for each of the main products was selected as the end point for the guidelines since it is in common to most end products, whereas secondary processing stages can vary greatly depending on the final product(s) produced.

Separate guidelines were developed in parallel for the animal feed supply chain (LEAP 2014d) and these covered emissions from all aspects of feed production through to the animal's mouth, whether the feed was produced on the livestock farm or purchased. Thus, it was important to align the LEAP Animal Feed Guidelines with those on the small ruminants (see Figure 2). Relevant points of alignment to avoid double counting or missed processes include 1) manure leaving a farm for application to feed crops leaves the system boundary of the small ruminants as it is collected (i.e. collection and on-farm storage emissions belong to the small ruminant guidelines), while all subsequent emissions from transport and after application are associated with feed production, and 2) any emissions associated with feed wastage at the animal feeding step on farm are accounted for in the small ruminant guidelines.

The main impact category covered by the small ruminant guidelines is climate change, estimated from the GHG emissions and expressed as CO_2-equivalents. Previous work done on guidelines for animal feeds provided the Feed TAG with the opportunity to expand the scope of the LEAP Animal Feed Guidelines to include other impact categories of eutrophication, acidification and land occupation. The GHG emissions in the small ruminant guidelines include methane (CH_4) emissions from rumen enteric fermentation and from animal dung and manure, nitrous oxide emissions (N_2O) from animal excreta deposited directly on grazing/browsing land and from manure collection and storage on-farm, and energy-related inputs from animal management, water supply and for primary processing (Figure 2). It also includes emissions from animal transportation, and primary processing emissions such as from the use of energy, consumables, refrigeration and wastes and waste-water processing.

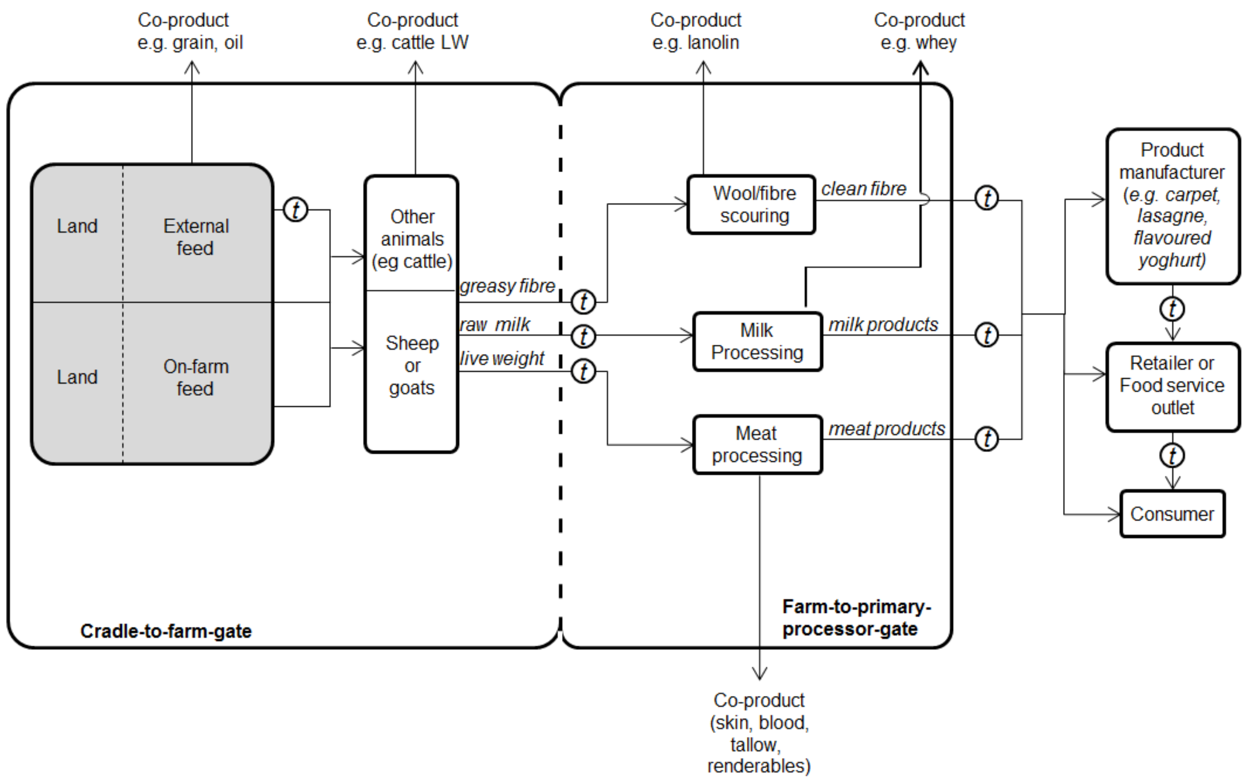

Figure 1. System boundary diagram for the cradle-to-primary-processing-gate for sheep and goat supply chains covering the three main products of fiber, milk and meat. The land and feed (inner shaded left box) is covered in the LEAP Animal Feeds Guidelines (LEAP 2014d). The ⓣ symbols refer to the main transportation stages.

The unit of analysis depends on the relevant stages of the life cycle covered. In view of the significance of the cradle-to-farm-gate stage, and that many published studies are limited to this component of the life cycle, these units have been expressed for this stage as well as for the cradle-to-primary-processing gate (Table 1). The equation for energy-corrected milk (ECM) is based on that from the IDF (2010) for dairy cow milk, to enable comparison between and within animal species, as:

kg ECM = kg milk x (0.1226 x fat% + 0.0776 x true-protein% + 0.0621 x lactose%)

Table 1. Recommended units of analysis for the three different main product types from small ruminants according to whether it is leaving the farm or primary product processing gate.

Main product type	Cradle-to-farm-gate	Cradle to primary-processing-gate
Meat	Live-weight (kg)	Meat product(s) (kg)
Fiber	Greasy weight (kg)	Clean weight (kg)
Milk	Energy-Corrected-Milk (kg)	Milk product(s) (kg)

The TAG was composed of eight selected experts drawn from six countries. Their backgrounds and complementary knowledge of products, systems and regions, allowed them to understand and address different interest groups and so ensure credible representation. While the TAG leader had a key role in drafting the guidelines, workshops and other interactions allowed consensus to be reached within the whole group. Before their release for public review, draft guidelines were reviewed by the LEAP Steering Committee and three independent reviewers.

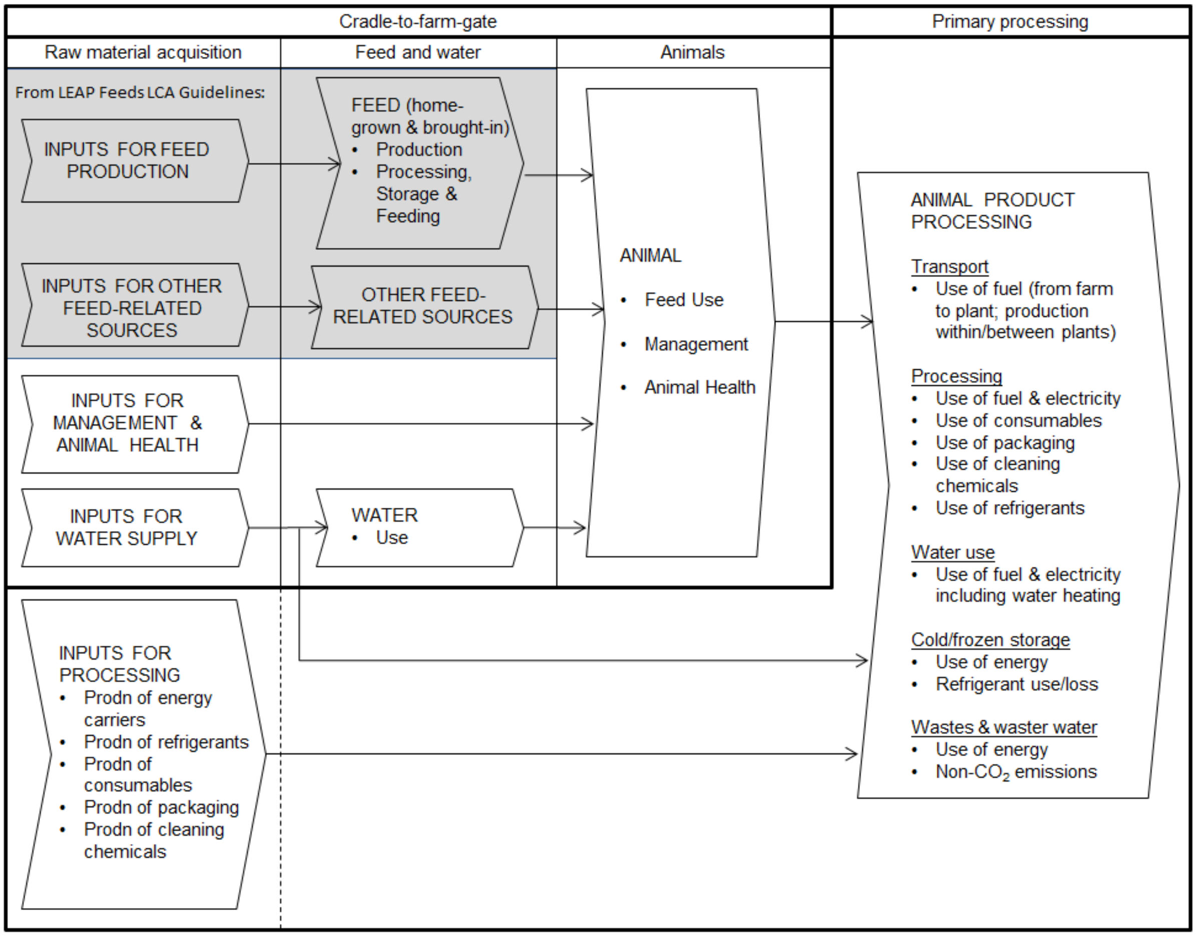

Figure 2. Processes that contribute to GHG emissions and fossil fuel demand within the system boundary of the cradle-to-primary-processing-gate for small ruminants. Note that the upper left shaded box refers to components covered by the LEAP Animal Feeds Guidelines (LEAP 2014d).

3. Results and Discussion

A methodology and guidelines for application were developed for determination of the GHG emissions and fossil energy demand from sheep or goat supply chains within the system boundary of the cradle-to-primary-processing-gate. The cradle-to-farm-gate stage was seen as being particularly important to cover in detail because of its significance to the whole life cycle. For example, Ledgard et al. (2011) evaluated the cradle-to-grave carbon footprint of lamb produced in New Zealand (NZ) and consumed in the United Kingdom and found that the cradle-to-farm-gate stage constituted 80% of total life cycle GHG emissions. The corresponding value for the primary meat processing stage was 3%.

Two areas of methodology identified as of key importance were 1) determining feed requirements by animals, and 2) the methods for handling multi-functional processes including allocation. These are discussed in the following sections.

3.1. Importance of determining feed requirements by animals

Within the cradle-to-farm-gate stage, the majority of GHG emissions from ruminant animals are determined by the amount of feed intake by animals, which is the main driver of enteric CH_4 emissions and of the amount of N excreted (the main source of N_2O emissions). Research in France and NZ (Gac et al. 2012; Ledgard et al. 2011) across diverse surveyed sheep farm systems showed that of total cradle-to-farm-gate GHG emissions, 53-73% were from enteric fermentation and 16-20% from excreta N_2O emissions. Thus, it was considered critical to obtain an accurate estimate of total feed intake and to use an IPCC tier-2 approach rather than a simple tier-1 approach. The latter is simply a constant value per animal and therefore would provide no opportunity to use the methodology to assess potential benefits from improved animal production and management practices.

In many small ruminant systems, much (or all) of the animal's time is spent grazing or browsing a range of forages. Thus, it is difficult to get a direct estimate of feed intake by primary data collection. Actual primary data on feed provided and consumed will often only be possible for the component of feed that might be stored on-farm or brought-in to the farm from an external source (e.g. concentrates). Consequently, an indirect modelling approach is required to calculate feed intake based on the energy requirements of animals. Most models used for calculation of feed requirements derive intake from the energy requirements for animal processes of growth, reproduction, fiber production, milk production, activity (i.e. grazing/walking) and maintenance (e.g. IPCC 2006; NRC 2007). The guidelines recommended that the choice of model/method should be based on a hierarchy of:

1. country-specific models used in the country National Greenhouse Gas Inventory;
2. other models that have been peer-reviewed and published that are appropriate to the region and country;
3. IPCC (2006) model;
4. IPCC default tier-1 values (this should be used as a last resort).

Associated data on the energy concentration of the feed(s) is required to convert energy intake by animals in feed(s) to dry matter intake. Again, this should be based on a hierarchy of: primary data for the specific feed type(s) where available; published values from the region or country for the specific feed type; or general published values (e.g. NRC 2007). Conversely, dry matter intake can be known or assessed from past research (from intake capacity by animals depending on level of productivity, e.g. as used by INRA, France) and energy intake can be derived from data on energy concentration of feeds.

Prior to calculating feed intake, the other critical aspect is to define the animal population and productivity over a representative one-year period. The animal population data must recognize the number of breeding animals (e.g. breeding ewes or does, rams or bucks, and the replacement breeding animals), as well as animals for production (e.g. surplus lambs or kids). This is illustrated for a simplified example of an animal population for a sheep farm system in Figure 3. From the base animal population data, an annual stock reconciliation needs to be derived that accounts for the time of lambing/kidding and time of sale of surplus animals. Ideally, a monthly or seasonal stock reconciliation would be used. The benefit of having a tier-2 methodology (using calculated energy requirements) and specific seasonal or monthly data is that the effects of improvement in animal productivity on reducing the carbon footprint of products can be determined. For example, achieving the final slaughter weight of lambs earlier results in a lower feed intake and the maintenance feed requirement is reduced relative to the feed needed to achieve a given level of animal production.

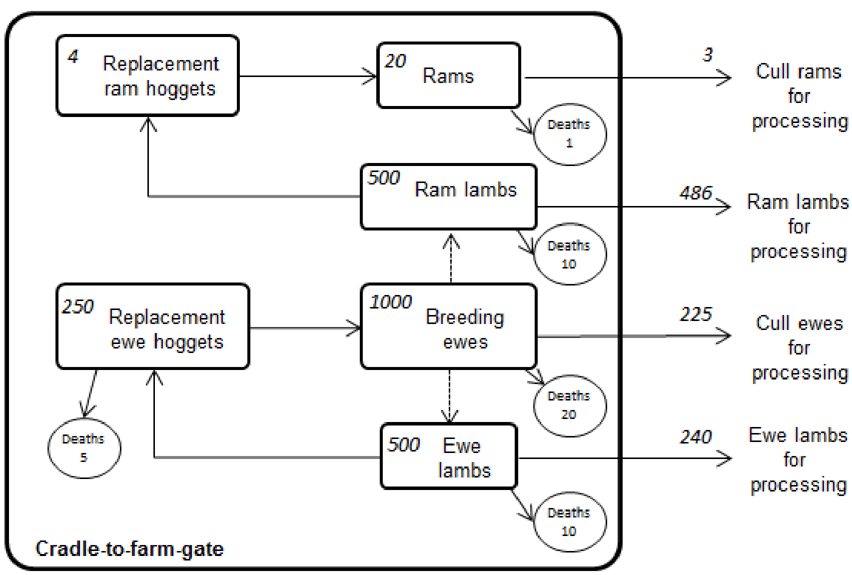

Figure 3: Simplified example of a sheep population illustrating relative numbers of breeding and replacement sheep on-farm and surplus sheep sold for meat processing (based on a breeding ewe flock of 1000, 100 percent lambing, 25 percent replacement rate, 2 percent death rate and first lambing at 2 years of age).

Calculation of animal productivity requires average data on male and female adult live-weight, live-weight of animal classes at slaughter, fiber production and milk production (for milking sheep or goats). Primary data on the animal population and productivity shall be used where possible.

3.2. Handling multi-functional processes including allocation of GHG emissions between co-products

There are a number of stages within the system boundary where multi-functional processes occur and where accounting for GHG emissions between different co-products is required (Figure 1). The ISO 14044 (2006) standard provided a hierarchy approach for accounting for co-products based on system separation wherever possible, followed by system expansion and then allocation options with an approach reflecting the underlying (bio)physical relationships between co-products being the preferred first option. As part of the LEAP process, a multi-functional output decision tree was developed (see LEAP 2014b,c,d) to further aid in selecting a system for handling co-products based on ISO 14044 (2006) and this was applied consistently across the small ruminant and poultry TAGs.

A summary of the recommended methods developed for handling co-products is given in Table 2, although it is also recommended that a sensitivity analysis involving several methods is applied to illustrate the effects of the choice of method used. Within the farm, there are two stages where choices relating to animal co-products may be required, i.e. for mixed species sharing the same feed resources, and for milk, meat and/or fiber production from sheep or goats (Figure 1). The preferred method for accounting for these multi-functional processes and co-products is firstly to separate activities related to the co-products where possible, and then to apply a biophysical approach based on energy intake from feed associated with the different animal species or co-products. This also recognizes that in ruminant livestock systems, the major determinant of GHG emissions within the farm stage is enteric CH_4 and excreta N_2O emissions, and the driver of these is feed intake (as noted in section 3.1).

Table 2. Recommended methods for dealing with multi-functional processes and allocation between co-products for the cradle-to-primary processing gate stages of the life cycle of small ruminant products

Source/stage of co-products	Recommended method*	Basis
Animal species (within farm)	1. Separate farm activities 2. Biophysical causality	First, separate the activities specific to an animal species and attribute the inputs/emissions accordingly. Then, determine emissions specific to feeds relating to the sheep or goats under study. If not possible and for remaining non-feed inputs, use biophysical allocation based on the proportion of total energy requirements for each of the different animal species.
Live-weight, fiber, milk for sheep or goats (at farm gate)	1. Separate activities 2. Biophysical causality	First, separate activities specific to products (e.g. electricity for shearing or milking). Then use biophysical allocation according to energy or protein requirements for animal physiological functions of growth, fiber production, milk production, reproduction, activity and maintenance.
Milk processing to milk products	1. Separate activities 2. Mass of fat + protein	First, separate activities specific to individual products where possible. Then use allocation based on the relative amount of fat + protein in the milk products
Fiber processing to clean fiber and lanolin	1. Separate activities 2. Economic	First, separate the activities specific to individual products where possible. Then use economic allocation based on a minimum of three years of recent average prices.
Meat processing to meat and non-meat products	1. Separate activities 2. Economic	First, separate the activities specific to individual products where possible. Then use economic allocation based a minimum of three years of recent average prices.

* *Note*: Where choice of allocation can have a significant effect on results, it is recommended to use more than one method to illustrate the effects of choice of allocation methodology

Defining a biophysical allocation methodology for handling co-products of milk, meat and fiber from sheep or goats at the farm-gate was most problematical, since this has received little research attention. For sheep or goats where milk is a main co-product, an allocation approach based on relative energy requirements for milk or meat production was considered most appropriate. This method also aligns with that agreed to in international guidelines for milk and meat from dairy cows by the IDF (2010). However, for animals where fiber is an important co-product this allocation approach was considered as less appropriate since fiber production is more commonly limited by protein rather than energy (e.g. CSIRO 2007). Thus, an approach based on use of a protein requirement model was considered as more desirable, but it was recognised that this is relatively untested. Recent studies by Wiedemann et al. (2014) indicated that biophysical allocation based on protein mass of the co-products can give similar results to that using a protein requirement modelling approach and that this may be a simpler methodology to use for allocation between fiber and meat.

An illustration of the effect of method of calculation for allocation between live-weight (for meat) and milk or fiber co-products at the farm gate is given in Table 3, based on use of data from Bett et al. (2007) for a Kenyan smallholder goat milk and meat production system, and from Ledgard et al. (2010) for a NZ hill country sheep production system. This shows relatively large effects of allocation method on the estimated % allocation values. It also shows the importance of applying a sensitivity analysis to illustrate the effects of choice of allocation method on the environmental footprint of co-products from small ruminants.

Table 3. Effect of allocation method on percentage allocation between live-weight (LW for meat) and milk or wool for case study goat or sheep systems

	Goats (Kenya) (Bett et al. 2007)	**Sheep (NZ) (Ledgard et al. 2010)**
Main products	Milk, LW for meat	LW for meat, wool (for carpet-making)
Farm type	Smallholder farm	Extensive-grazing pasture farm
Percent allocation to milk (goats) or wool (sheep):		
Biophysical – energy requirements	70%	16%
Biophysical – protein requirements	n.d.	37%
Biophysical – protein mass	50%	39%
Economic	45%	19%

During the primary processing stage, there can also be multiple co-products depending on the type of main products being processed. As part of the multi-functional output decision tree it was defined that a biophysical allocation approach is appropriate where co-products have similar physical properties and serve similar goals or markets, but that where this does not apply the use of economic allocation is appropriate. Based on these criteria, biophysical allocation is recommended for dealing with different milk products, while economic allocation was identified for handling co-products from fiber and meat processing since the latter produce secondary products for very different end uses. For milk products, fat and protein are the key constituents in the co-products and therefore allocation based on the mass of fat plus protein is recommended. This aligns with recent approaches described for dairy cow milk products (Flysjö 2011; Thoma et al. 2013).

Primary processing of greasy fiber from sheep or goats can result in clean wool, lanolin and residue (vegetable matter and dirt, which usually goes to waste). The latter residue is often a valueless waste, but in some cases it may be further processed to a valuable conditioner or fertilizer, and in such cases it should be treated as a co-product. Economic allocation (based on a recent average of a minimum of three years data to reduce effects of temporal variability) of co-products from primary processing of fiber is recommended. In practice, the recovery of lanolin from greasy wool of sheep amounts to only about 2-7 percent by weight (higher for finer wool and lower for goat fibers) or possibly slightly higher on an economic basis, and therefore most of the resource use and GHG emissions will be allocated to the fiber.

Primary processing of goats and sheep for meat production can result in a wide range of co-products, including hides (for leather), tallow (e.g. for soap, biofuel), pet food, blood (e.g. for pharmaceutical products), fiber and renderable material (e.g. for fertilizer). Also, the proportion and components of the carcass used for meat products for human consumption and for co-products is dependent on culture and economic factors and can differ significantly. In view of the very varied use of these co-products and the protocol relating to the multi-functional output decision tree, economic allocation methodology is appropriate. However, for the various edible meat products, it is recommended that they all be treated the same per-kg (i.e. no separate economic allocation between meat cuts). An example of the differences in weight and relative economic value of different meat cuts and non-meat co-products is given in Table 4 for the average lamb from NZ abattoirs in mid-2009. It shows that there was more than an eight-fold difference in price per kg between the lamb rack and neck cuts of meat and illustrates the potential effect that application of economic allocation between meat cuts could have had. Use of data from Table 4 results in calculated values for the % allocation for meat relative to that for meat plus other non-meat co-products at 88% based on economic allocation and 51% based on mass allocation. This difference reflects the greater economic value of meat than of the non-meat co-products.

Table 4. Variation in the mass and economic value of components of an average New Zealand lamb leaving an abattoir and the effects on allocation calculations (data provided by NZ Meat Industry Association from 2009)

	Average mass of component (kg)	Component as a % of total mass	Price per-kg relative to leg meat	Component as % of total economic value
Meat:				
Neck	0.54	1.5	0.21	0.8
Shoulder	4.6	12.7	0.51	16.1
Rack	1.21	3.4	1.73	14.3
Breast and shank	1.46	4.1	0.47	4.8
Loin	1.43	4.0	1.04	10.2
Legs	4.68	13.0	1.00	32.1
Other meat	2.43	6.7	0.38	6.4
Edible offal	2.0	5.5	0.28	3.9
Co-products:				
Hide/skin	2.21	6.1	0.28	4.3
Wool	1.59	4.4	0.27	3.0
Blood	1.76	4.9	0.01	0.1
Inedible offal	0.65	1.8	0.14	0.6
Rendering/tallow	11.54	32.0	0.04	3.5

4. Conclusions

Key learnings from development of the guidelines included the need to recognize and account for 1) the diversity of small ruminant production and processing systems, 2) the potential for limited data availability, and 3) the importance of several main areas of methodology in determining the final results. The LEAP partnership offered an effective process for stakeholders in the chain to overcome these issues in a concerted manner. The guidelines were structured so that they could be applied to a wide range of production and processing systems using the generic principles and varying levels of specification provided. Additionally, for several processes a hierarchy approach was presented that identified the most appropriate method or data source as well as alternative options that were less detailed or less specific but more achievable for systems with limited data.

Two critical areas of methodology identified were the importance of determining feed requirements by animals and the methods for handling multi-functional processes and allocation. Relatively large sections of the guidelines were dedicated to these areas of methodology, by providing greater specificity on data sources and methods of calculation. In this paper, case study examples were given to show the effects of the methodology choices for co-product handling including allocation options. It was concluded that sensitivity analysis should be carried out and results presented to illustrate the effects of these key methodology choices on the calculated environmental emissions.

An additional learning from development of the draft guidelines for small ruminants concurrently with those for feed and poultry was the importance of harmonization across stages of the supply chain (feed and animal production) and between species. Further adjustments of the Guidelines are expected based on the on-going Public Review and a planned test phase. It is also LEAP's aim to expand the guidelines to the inclusion of other environmental impact categories. This collaborative approach, which is also currently being applied to other species (e.g. large ruminants), has potential to improve consistency in assessing and reporting the environmental performance of livestock and in monitoring improvements.

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Comparing UK turkey production systems using analytical error propagation in uncertainty analysis

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ABSTRACT

The aim of this study was to quantify the environmental impacts per unit of live weight of the main UK turkey systems, namely stag and hen production with either controlled or natural ventilation. An LCA modeling framework, based on system approach and mechanistic sub-models was applied for this purpose. For the first time, detailed production data from the industry was used as input. The differences between the systems were analyzed using an analytical “top-down” method for uncertainty analysis, developed specifically for this study. The results show that there were only small, mainly non-significant differences in the impacts between the systems, affected mainly by their feed conversion ratio and slaughter weight. The novel uncertainty analysis method makes it possible to obtain exact quantification of the overall uncertainties of outputs of complex systems models without the need of time consuming and approximate Monte Carlo simulations.

Keywords: acidification potential, eutrophication potential, global warming potential, turkey production, uncertainty analysis

1. Introduction

According to Defra (2011), the UK turkey production in 2010 was 162 thousand tons carcass weight, which is over 10% of the total UK poultry meat production. Despite the importance of turkey systems and their potential contribution to the overall environmental consequences of livestock production, the environmental impacts of turkey production have so far only been analyzed using generic UK data (Williams et al. 2006). In order to quantify the current impacts of the main turkey systems, and to find ways to improve their environmental friendliness, detailed production data from the main turkey production companies is required, combined with a systematic assessment methods, such as agricultural LCA.

In earlier studies (Leinonen et al. 2012a; b; Williams et al. 2006), a systems approach-based modeling framework was developed to quantify the environmental impacts of poultry systems. As an essential part of this framework, a method for uncertainty analysis was also developed, in order to make statistical comparisons between the systems under consideration. In these studies, the uncertainty was quantified using Monte Carlo simulations, which is a common practice in agricultural LCA. However, the problem with such an approach is that it requires a lot of computing power, it is time consuming, and the output is only an approximation, the accuracy of which depends on the number of Monte Carlo runs used. Therefore, it is quite clear that there is a need for a more straightforward, preferably analytical method for quantifying the uncertainty in agricultural LCA studies.

The aim of the current study was to apply the LCA method, “from cradle to gate” to quantify and compare the environmental burdens of the main production systems in UK, namely 1) Stags (males) with controlled ventilation, 2) Hens (females) with controlled ventilation, 3) Stags with natural ventilation, and 4) Hens with natural ventilation. Another aim was to develop a novel analytical method for uncertainty analysis, which would allow exact quantification of the system uncertainties, without time consuming Monte Carlo simulations which could give only an approximate solution.

2. Methods

2.1. Systems approach and the data

The general approach taken in the current study was with systems modeling of production. This included structural models of the industry, process models and simulation models that were unified in the systems approach so that changes in one area caused consistent interactions elsewhere. This approach was applied to both feed crop and animal production. The systems modeled in this study included crop production, non-crop nutrient production, feed processing, breeding, turkey brooding, turkey finishing and manure and general waste management, following the overall principles presented by Williams et al. (2006) and Leinonen et al. (2012a;b).

The production systems in this study were considered to represent typical mainstream UK turkey production, i.e. stags and hens, both with either controlled or natural ventilation. The farm energy consumption for heating, lighting, ventilation and feeding was based on average data from typical farms as provided by the main UK turkey production companies. Information about the type and amount of bedding and other material use was also obtained from the industry. The bird performance and production data, including the length of the production cycle, stocking density, final weight, feed intake and mortality came from actual farm data provided by the industry. The main production figures for different systems are presented in Table 1. Additional data, such as life cycle inventories (LCI) of agricultural buildings and machinery, came from Williams et al. (2006). The baseline diets representative of those used in the UK for each system were constructed using information provided by the turkey industry, and the environmental impacts arising from feed production were calculated based on the relative proportion of each ingredient in these diets.

Table 1. The average production figures for the main UK turkey systems considered in this study

System	Age at slaughter, days	Weight at slaughter, kg	Feed conversion ratio, kg feed/kg live weight
Stags, controlled ventilation[1]	133	15.2	3.1
Hens, controlled ventilation	95	7.3	2.8
Stags, natural ventilation	137	16.6	3.1
Hens, natural ventilation	90	6.7	2.6

[1]In this system, the birds were predominantly males, but some farms with mixed flocks with small amount of females are also included.

2.2. The models

The structural model for turkey systems calculated all of the inputs required to produce the functional unit (1000 kg of live weight), allowing for breeding overheads, mortalities and productivity levels. It also calculated the outputs, both useful and unwanted. Changes in the proportion of any activity resulted in changes to the proportions of others in order to keep producing the desired amount of output. Establishing how much of each activity was required was found by solving linear equations that described the relationships that linked the activities together.

The model calculated the N, P and K contents of the manure according to the mass balance principle, i.e. the nutrients retained in the animal body were subtracted from the total amount of nutrients obtained from the feed. For the purpose of the study, it was assumed that all manure was transported for soil improvement.

A separate sub-model for arable production was used to quantify the environmental impacts of the main feed ingredients, with main features as in Williams et al. (2010). All major crops used for production of turkey feed were modeled. For the crops produced overseas (soya, palm oil) the production was modeled as closely as possible using local techniques, and transport burdens for importing were also included. The greenhouse gas emissions arising from land use change were taken into account according to the principles of the carbon footprinting method PAS 2050:2011 (BSI 2011).

A separate sub-model was also used for manure and the nutrient cycle. In the model, the main nutrients that were applied to the soil in manure were accounted for as either crop products or as losses to the environment. N was explicitly partitioned into readily plant available and slow-release organic N. The crop yield response was compared with that from manufactured N and the benefits were credited to poultry by offsetting the need to apply fertilizers to winter wheat as described by Sandars et al. (2003) and implemented by Williams et al. (2006). Losses as nitrate or by denitrification were calculated on a long term basis.

2.3. Environmental impacts

Emissions to the environment were aggregated into environmentally functional groups as follows. Global Warming Potential (GWP) was calculated using a timescale of 100 years. The main sources of GWP in turkey industry are carbon dioxide (CO_2) from fossil fuel, nitrous oxide (N_2O) and methane (CH_4). GWP was quantified as CO_2 equivalent: with a 100 year timescale 1 kg CH_4 and N_2O are equivalent to 25 and 298 kg CO_2 respectively (Foster et al. 2007).

Eutrophication Potential (EP) was calculated using the method of the Institute of Environmental Sciences (CML) at Leiden University (http://www.leidenuniv.nl/interfac/cml/ssp/index.html). The main sources in turkey production are nitrate (NO_3^-) and phosphate (PO_4^{3-}) leaching to water and ammonia (NH_3) emissions to air. EP was quantified in terms of phosphate equivalents: 1 kg NO_3-N and NH_3-N are equivalent to 0.44 and 0.43 kg PO_4^{3-}, respectively.

Acidification Potential (AP) was also calculated using the method of the Institute of Environmental Sciences (CML) at Leiden University. The main source in turkey industry is ammonia emissions, together with sulfur dioxide (SO_2) from fossil fuel combustion. Ammonia contributes to AP despite being alkaline; when emitted into the atmosphere, it is oxidized to nitric acid. AP was quantified in terms of SO_2 equivalents: 1 kg NH_3-N is equivalent to 2.3 kg SO_2.

Primary Energy Use included all the energy needed for extraction and supply of energy carriers.

2.4. Breakdown of the environmental impacts

The results were broken down by the following material (and energy) flow categories (or sub-systems) to demonstrate their relative contribution to the overall impacts:

1) Feed: production of crops and additives, feed processing and transport. This category also includes the water consumed during housing.

2) Farm Electricity: direct electricity consumption at the farms (breeding, brooding and finishing) and hatcheries, not including feed production, processing and transport.

3) Farm Gas and Oil: direct fuel consumption at the farms and hatcheries, not including feed production, processing and transport.

4) Housing: direct emissions of NH_3, CH_4 and N_2O from housing and burdens from construction of farm buildings and vehicles, not including buildings and vehicles used in feed production, processing and transport of ingredients.

5) Manure and Bedding: emissions from manure storage and field spreading and the production of the bedding. This category also includes credits from replacing synthetic fertilizers. It does not include direct emissions of NH_3, CH_4 and N_2O from housing.

2.5. Uncertainty analysis

The uncertainties in the input variables can be divided into two groups, namely “alpha” and “beta” uncertainties, according to a concept first presented by Wiltshire et al. (2009) and Chatterton et al. (2010), and later applied for LCA for poultry production by Leinonen et al. (2012a). Alpha uncertainties can be considered to vary between systems, and therefore they should be taken into account in statistical analyses of the differences between the systems. For example, variation between farms in production, feed intake and energy use figures can all be considered to represent alpha uncertainties. In contrast, beta uncertainties are considered to be similar between the systems, and, following the principles of earlier studies, they were assumed to have no effect on the statistical comparison between the systems (Wiltshire et al. 2009; Chatterton et al. 2010; Leinonen et al. 2012a). Examples of the beta uncertainties are the emission factor for N_2O from manure, conversion factor from electricity to primary energy, and in general errors related to the modeling framework. In this study, the aim was to compare statistically the differences between production systems, and therefore only the alpha uncertainties are considered in the following.

To quantify the overall alpha uncertainty of the outputs of the LCA model, a novel, analytical “top-down” approach was developed for this study. In this analysis, the outputs (i.e. each category of environmental impacts per functional unit) were divided into separate components based on their sources, e.g. feed production, manure management and on-farm electricity use. Each component (*i*) was then expressed as a simplified function, and the total emissions of the system were calculated as their sum. For example for GWP, this can be expressed as:

$$GWP = \sum EC_i \times Activity_i \qquad \text{Eq. 1}$$

where $Activity_i$ represents for example material or energy use per FU, and the “emission coefficient” EC_i is the quantity of greenhouse gas (GHG) emissions per unit of activity. Depending on the context, the emission coefficient can be a simple emission factor, result of a complex function or outcome of the LCA model. In some cases, the EC_i can be considered to have both alpha and beta uncertainties, and in some other cases beta uncertainties only. If the latter option is true, then the uncertainty of the emissions coefficient can be ignored in the uncertainty analysis.

For example in the case of GWP, the emissions arising from feed production and manure management can be expressed with a simple linear function where the "activity" represents the feed consumption per functional unit, i.e. the feed conversion ratio (FCR):

$$F + M = a + (EC_f + EC_m) \times FCR \qquad \text{Eq. 2}$$

where F and M are the GHG emissions from feed production and manure, respectively, EC_f and EC_m are the emissions coefficients (calculated by fitting the above equation to the outputs of the full LCA model) for feed and manure, respectively, and FCR is the average feed conversion ratio, i.e. the mass of feed divided by the liveweight gain. The constant a, which was also calculated from the model outputs, is needed in the equation because the manure emissions are not directly proportional to the feed intake. Instead, it can be considered that there is a theoretical, (very low) level of feed intake where the nutrient intake equals the nutrient retention (assuming that this remains unchanged), and as a result, the emissions from manure are zero at that point.

Now the alpha uncertainty for the GHG emissions from feed production and manure management can be quantified using a general uncertainty propagation rule (e.g. Taylor 1996):

$$\left(\frac{\sigma_{F+M}}{F+M}\right)^2 = \left(\frac{\sigma_{EC_f}}{EC_f + EC_m}\right)^2 + \left(\frac{\sigma_{FCR}}{FCR}\right)^2 \qquad \text{Eq. 3}$$

where σ_{F+M} is the alpha uncertainty (standard deviation) of the combined feed and manure emissions, σ_{ECf} the alpha uncertainty of the feed production emissions (quantified from the output of the crop production sub-model), and σ_{FCR} the variation in the feed conversion ratio, quantified directly from the data provided by the turkey industry. The alpha uncertainty for the manure emission coefficient was considered to be zero and therefore it does not appear in the equation.

For the other GHG emissions, the emissions from direct farm electricity use (E) can be expressed simply as:

$$E = EC_e \times Eu \qquad \text{Eq. 4}$$

where EC_f is the emission coefficient for electricity and Eu is the average farm electricity use per functional unit. The emissions for farm gas consumption (G), or other fuel consumption where applicable, can be expressed exactly in the same way.

Again, the alpha uncertainty for the emissions from direct electricity use (and similarly for direct gas and other fuel use) can be quantified as:

$$\sigma_E^2 = EC_e^2 \times \sigma_{Eu}^2 \qquad \text{Eq. 5}$$

where the variation in the electricity consumption per functional unit (σ_{Ec}) comes directly from the industry data.

The direct GHG emissions from housing (H) are mainly N_2O in turkey production, and they were not included in the manure emissions described above. Instead, the housing N_2O emissions were modeled separately based on the live weight and the duration of housing. Therefore, these emissions per live weight can be expressed as:

$$H = EC_h \times Age \qquad \text{Eq. 6}$$

where Age is the average slaughter age of the birds. Now the alpha uncertainties for the housing emissions can be calculated as:

$$\sigma_H^2 = EC_h^2 \times \sigma_{Age}^2 \qquad \text{Eq. 7}$$

where the variation of the slaughter age (σ_{Age}) comes directly from the industry data.

Finally, the overall alpha uncertainty for the total GHG emissions can be quantified as sum of the alpha uncertainties of the different components presented above:

$$\sigma_{GWP}^2 = \sigma_{F+M}^2 + \sigma_H^2 + \sigma_E^2 + \sigma_G^2 \qquad \text{Eq. 8}$$

It should also be noted that when the uncertainty terms of the above equations are correlated, their covariance should also be taken into account in error propagation. However, in the case of the turkey emission model, most of the activities considered can be assumed to be independent from each other. The only exceptions are the slaughter age (used to model the emissions from housing) and the feed conversion ratio (feed and manure emissions). In theory, FCR should increase with the increasing age of the bird. However, in the data used in the present study, no consistent correlation between these variables was found within the systems, so the covariance between these variables was ignored in Eq. 8.

The uncertainties for other impact categories could be calculated mainly following the principles presented above. The main difference can be found in Eutrophication and Acidification Potentials, where the combined emission from feed production, housing and manure management can be expressed as a linear function of FCR. For example, for Eutrophication Potential this can be expressed as follows:

$$F + H + M = a + (EC_f + EC_h + EC_m) \times FCR \qquad \text{Eq. 9}$$

where F, H and M are the components of Eutrophication potential related to feed production, housing and manure management, respectively, EC_f , EC_h and EC_m are the eutrophication emission coefficients for feed, housing and manure, respectively. The alpha uncertainty for the EP arising from these components can be then quantified as:

$$\left(\frac{\sigma_{F+H+M}}{F+H+M}\right)^2 = \left(\frac{\sigma_{EC_f}}{EC_f+EC_h+EC_m}\right)^2 + \left(\frac{\sigma_{FCR}}{FCR}\right)^2 \qquad \text{Eq. 10}$$

Finally, a statistical analysis was conducted to evaluate the differences (at the P < 0.05 level) between the systems, based on the overall alpha uncertainties of each impact category, as described by Leinonen et al. (2012a).

3. Results

The environmental impacts of different turkey production systems, the contribution of different subsystems, and the alpha uncertainties of the impacts are presented in Figures 1-4. The results show that feed production, processing and transport was the main source of the impacts in the categories of Primary Energy Use and Global Warming Potential, while emissions from manure management had the biggest contribution to Eutrophication and Acidification Potentials. There was a lot of variation in the direct farm energy use (especially gas for heating) between the systems, but this had a relative small contribution to the overall impacts.

When the overall alpha uncertainties were applied in the statistical comparison between the systems, it was found that in most environmental impact categories there were generally no significant differences (P < 0.05) in the environmental impacts between the systems. A significant difference was found in Acidification Potential, where the stag system with controlled ventilation had a higher impact than the hen system with natural ventilation.

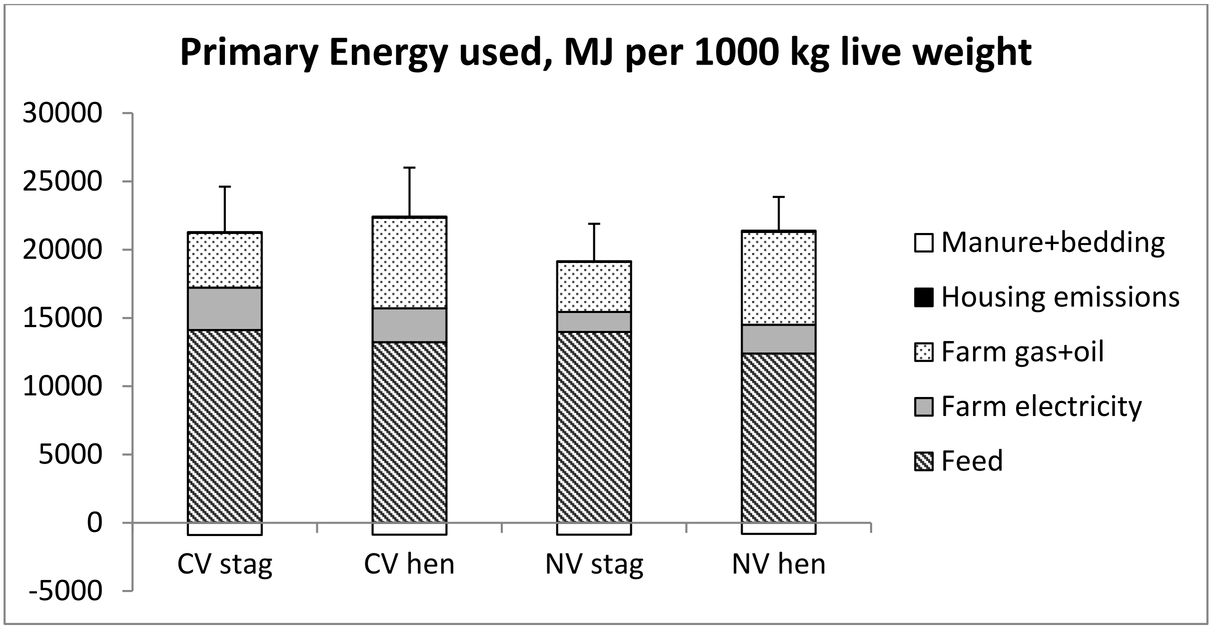

Figure 1. The Primary Energy Use of the four main UK turkey production systems. The error bars indicate standard deviations based on alpha uncertainties. CV=controlled ventilation, NV=natural ventilation.

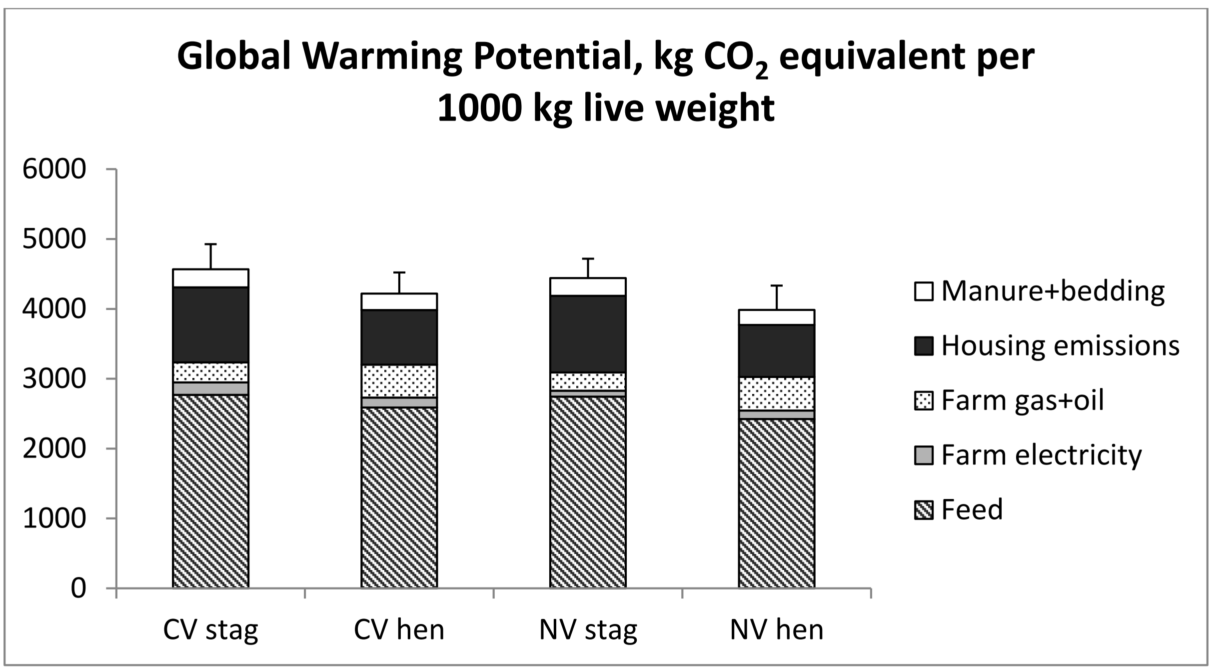

Figure 2. The Global Warming Potential of the four main UK turkey production systems. The error bars indicate standard deviations based on alpha uncertainties. CV = controlled ventilation, NV = natural ventilation.

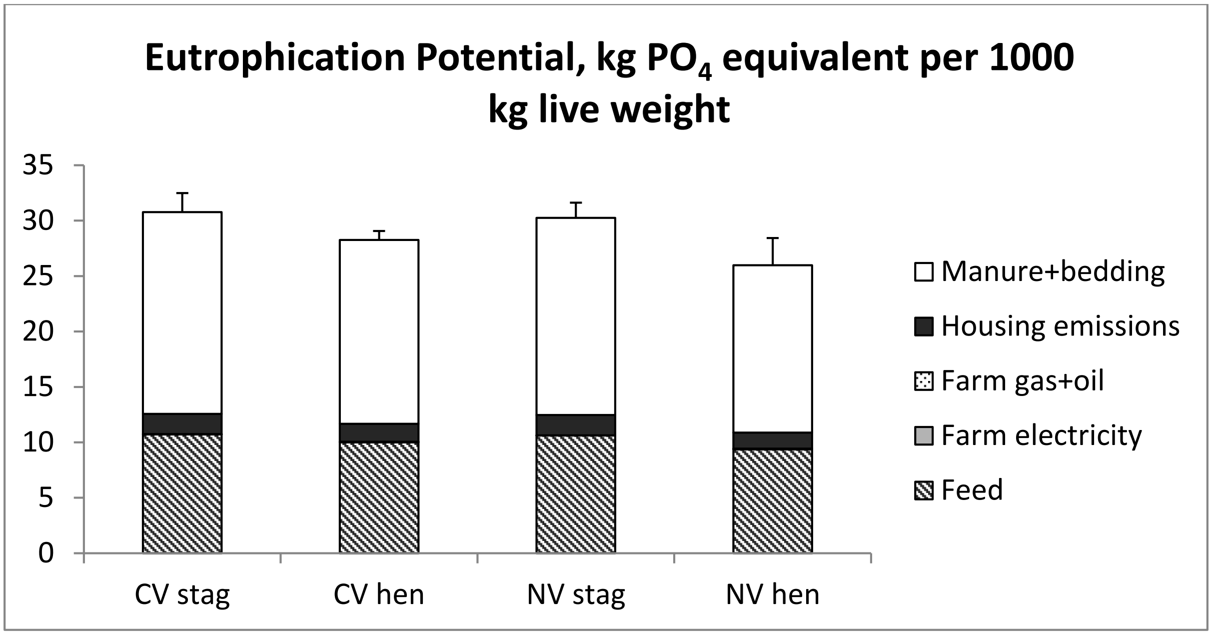

Figure 3. The Eutrophication Potential of the four main UK turkey production systems. The error bars indicate standard deviations based on alpha uncertainties. CV = controlled ventilation, NV = natural ventilation.

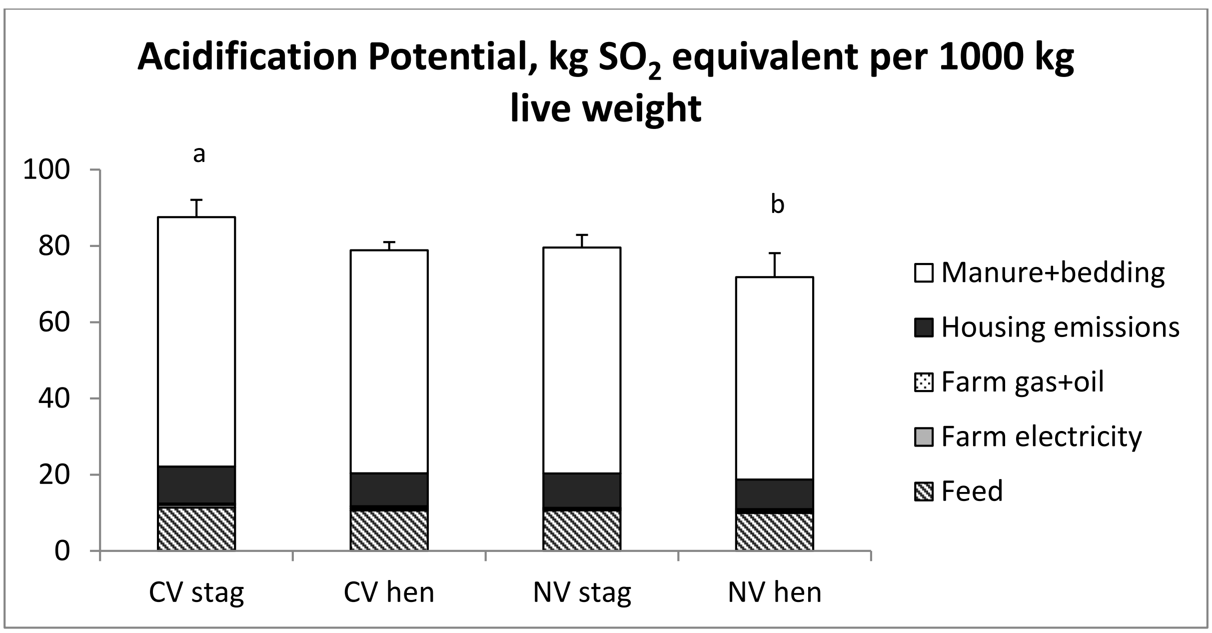

Figure 4. The Acidification Potential of the four main UK turkey production systems. The error bars indicate standard deviations based on alpha uncertainties. Different letters indicate a statistically significant ($P<0.05$) difference between the systems. CV = controlled ventilation, NV = natural ventilation.

4. Discussion

There were only small (mainly non-significant) differences in the environmental impacts between UK turkey production systems, although some systematic trends occurred in certain sources of the impacts. These trends were mainly related to feed conversion ratio, which affected the emissions from feed production, housing and manure management. When the cycle length and the final body weight increase, an increase in the feed intake per unit of live weight is also expected, as the relative contribution of the energy required for maintenance, as opposed to growth, increases with the age of the bird. In the systems compared in this study, the hen systems always had a shorter cycle length and lower body weight compared to the stag systems. For this reason, the FCR was lower for hens than for stags, and the hen systems consequently tended to have lower impacts than the stag systems, especially for the Eutrophication and Acidification potentials.

In contrast, the farm energy consumption, especially heating gas, was higher per unit of live weight for the hen systems than for the stag systems. This difference was again related to lower slaughter weight of hens. Although it can be expected that for example the heating requirement would be similar per bird for stags and hens (as heating is mainly required for young birds only), hens produce less output (i.e. live weight) per bird, and therefore the impact per the functional unit is higher. Thus the better yield in the stag systems partially compensated the higher impacts related to poorer feed conversion in some impact categories. This leads to the observation that mitigation methods, such as using renewable energy for heating young birds would be more effective for hens than stags.

Any quantitative comparison between different systems is not feasible if the range of uncertainty in the results is not available. Despite this well-known fact, uncertainty analysis is not commonly applied in LCA studies for agricultural products. This omission can be partly explained by the extensive requirement of the data, in order to quantify the uncertainties in the input variables reliably. However, another reason might be the limitations of the chosen calculation method itself. Monte Carlo simulations are usually used in uncertainty analysis. Hence, the model used to quantify the environmental impacts is run several (generally thousands) times, and for each run, the input parameters are randomly sampled from a pre-determined distribution. Running of very complex models thousands of times requires a lot of computing power, and is also time consuming. Furthermore, the accuracy of the outputs of the Monte Carlo simulations (e.g. estimates for means or standard deviations) is directly dependent on the number of the runs. Therefore, the results of the Monte Carlo method can be seen as compromise between the reliability of the results and the use of computing power.

The problems related to the use Monte Carlo simulations in uncertainty analysis could be, in theory, avoided if the error propagation within the systems under consideration could be done analytically. The problem here is, however, that with very complex mathematical models, any analytical solution of uncertainty would require a huge amount of complicated calculations and in many cases could be impossible to perform. In the present study, a potential solution for this problem is presented by applying a "top-down" method, where the outputs of a complicated model are expressed with simple functions of the driving "Activity" variables. The contributions of all complicated sub-models are thus aggregated into simple emission coefficients. Quantifying the values of these coefficients would require running the model, in some cases several times (in the case of linear relationships, two model runs is enough), and then examining the broken-down outputs of the model to specify the relationships between the "Activities" and the "Emission coefficients".

Despite its apparent complexity, the approach presented here can save much computing time compared to the thousands Monte Carlo runs, and if done correctly, it also provides an exact solution as opposite to the Monte Carlo approximations. A potential limitation of this analytical method is that in its current form it assumes that all uncertainties are normally distributed, while the Monte Carlo runs can utilize any type of distribution. Although the effect of this assumption was not examined in this study, it can be expected to have only minor effects on the results, keeping in mind that in many cases the actual form of the distribution of the input data may not be very well known anyway. So it can be expected that the overall principles of the analytical uncertainty analysis presented in this study would be applicable and beneficial in several applications of agricultural LCA, especially if their purpose is to perform quantitative comparison between separate systems or scenarios.

5. Conclusion

This study presents the environmental impacts of main UK turkey production systems, quantified for the first time using detailed industry data. Furthermore, a novel approach to uncertainty analysis is presented, where the "alpha" uncertainties, i.e. those varying between the systems under comparison are quantified analytically, thus saving time and computing power generally related to uncertainty analyses carried out with Monte Carlo simulations. The results of the system comparison show that in general there are only small, non-significant differences in the impacts between different systems. The main system-related variables affecting the impacts are the feed conversion ratio, which affects the food, housing and manure emissions, and the slaughter weight, affecting the energy use per functional unit.

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Evaluating methods to account for the greenhouse gas emissions from Land Use Changes in agricultural LCA

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ABSTRACT

Approaches to account for land use change-related GHG emissions (LUCE), differing on the basis of the spatial allocation of the emissions, were compared in this study, and the justification of their application was evaluated based on generally accepted criteria from ISO 14040, PAS 2050 and IPCC guidelines. In general, most methods technically fulfilled those criteria, with exception of the "Worst case" or marginal approach which resulted in multiple counting of global emissions. The selection amongst the acceptable methods still remains rather subjective. The conclusions of agricultural LCA studies, related for example to the GHG emissions from livestock feed, are strongly dependent on the selected method, and for this reason a universally accepted single method is needed. One solution might be an application of a modified "Top down" method, which takes into account emissions from indirect LUC and the actual drivers affecting them.

Keywords: global warming potential, greenhouse gas emissions, land use change, livestock feed, soy

1. Introduction

It is becoming increasingly evident in agricultural LCA, especially of livestock production, that selecting the method for accounting for the greenhouse gas (GHG) emissions from land use change (LUCE) is probably the most critical step of the assessment (e.g. Leinonen et al. 2013; Middelaar et al. 2013). At present, there are several different approaches to LUCE, which may give highly varied estimates for the emissions for different agricultural crops, and therefore may have strong impacts on the conclusions of the study in question. Therefore, it is obvious that some kind of agreement on the LUCE accounting methodology is needed, and the criteria for selecting the method should be clearly specified.

Most of the recent LCA studies on agricultural production have aimed to fulfil the requirements of the ISO 14040 standard (BSI 2006), the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006), and more recently, also the more detailed methodological criteria described in the Publicly Available Specification PAS 2050:2011 (BSI 2011). As a result, it could be expected that the methods quantifying the GHG emissions arising from land use change (LUC) should follow the same criteria.

Some common principles required for the methods accounting for LUCE can be identified from the three documents mentioned above. One of such principles is that when quantifying existing LUC emissions, an attributional approach should be applied, meaning "*LCI modelling frame that inventories the inputs and output flows of all processes of a system as they occur*", as defined in ILCD (2010) . This requirement is the one of the overall principles of PAS 2050:2011, and it is stated there as "*unless otherwise indicated, the assessment of the life cycle GHG emissions of products shall be made using the attributional approach, i.e. by describing the inputs and their associated emissions attributed to the delivery of a specified amount of the product functional unit*". Similarly, as the IPCC guidelines aim to quantification of the national GHG emissions, they also concentrate on the existing state, i.e. apply attributional approach as opposed to consequential, which would be applicable for scenarios of potential changes in the future.

A logical consequence of the attributional approach is that the analysis must fulfil the criteria of mass and energy conservation. This is directly stated in the ISO 14040 as "*As each unit process obeys the laws of conservation of mass and energy, mass and energy balances provide a useful check on the validity of a unit process description*". Also, the IPCC Guidelines recommend checking the mass balance of the emissions, "*to avoid omissions or double counting*".

The requirement of mass conservation is also related to some of the criteria of PAS 2050. These criteria include the following:

1) Completeness: "*all product life cycle GHG emissions and removals arising within the system and temporal boundaries for a specified product which provide a material contribution to the assessment of GHG emissions arising from that product have been included*", and

2) Consistency: "*assumptions, methods and data have been applied in the same way throughout the quantification and support reproducible, comparable outcomes*".

In the following, some generally used methods for accounting for LUC-related emissions in LCA studies are considered, the justification of their use is evaluated in terms of the criteria specified above, and their consequences on the application in agricultural LCA are demonstrated. In the latter, the Global Warming Potential of broiler feed is used as an example.

2. Methods

2.1. Alternative approaches to LUCE accounting

In an ideal situation, the net emissions of greenhouse gases from a certain land area, arising as a result of agricultural LUC, should be quantified and allocated to crops that are grown within this area during the time period when the LUC-related emissions are considered to occur. In PAS 2050:2011, this is the preferred situation and is defined as "*where the country of production is known and the previous land use is known, the GHG emissions and removals arising from land use change shall be those resulting from the change in land use from the previous land use to the current land use in that country*". In this paper, this approach is referred as the "Actual LUC scenario". In practice, this approach could be applied to crops originating from a country where no LUC is known to occur, or alternatively to crops that come from a country where LUC may have occurred, but that specific crop is certified to originate from "mature" agricultural land, i.e. land from which no more LUC emissions are assumed to arise. This stage is generally considered to be reached after a selected amortization period (typically 20 years).

In several practical situations, it is not possible to know the exact origin of a certain agricultural commodity, and therefore it is not known whether it originates from land recently converted to agricultural use and whether the LUC-related emissions should be included in the carbon footprint of that product. A solution for this is to quantify the overall LUC emissions from the production of a certain crop within a bigger area, e.g. a country or a region, and then allocate the emissions evenly to that crop, produced at any location within that area. PAS 2050:2011 gives two options for that approach. First, it is stated that "*where the country of production is known, but the former land use is not known, the GHG emissions arising from land use change shall be the estimate of average emissions from the land use change for that crop in that country*". Second, an even more general option in PAS 2050:2011 says that "*where neither the country of production nor the former land use is known, the GHG emissions arising from land use change shall be the weighted average of the average land use change emissions of that commodity in the countries in which it is grown*". In this paper, this approach is referred as the "Best estimate scenario".

An alternative approach to account for the LUC-related emissions considers both the direct and indirect LUC related to crop production. It can be argued that all agricultural activity has indirect LUC effects, i.e. growing more of any crop in any economically-connected location in order to meet global demand will increase the land use pressure elsewhere (although the opposite could happen in the case of intensification of the production). Thus, the global LUC emissions should be equally allocated to all crops per ha, regardless their actual location (Audsley et al. 2009). According to this scenario, equal LUC emissions of 1430 kg CO_2 ha^{-1} y^{-1} should be included in the production of all crops, regardless the country of their origin or the previous land use (Audsley et al. 2009). The GWP from LUC per kg of each feed ingredient is thus dependent on the land area required for its production, so for example crops with high yield per ha have lower LUC emissions than crops with low yield. There is also no distinction in the attribution of LUCE to crops with high or low rates of expansion. In this paper, this approach is referred as the "Top-down" scenario"

The main difference between the approaches described above is how the GHG emissions arising from LUC are allocated spatially, ranging from the level of a single field of a specific crop ("Actual LUC" scenario) to allocation evenly to all agricultural land ("Top-down" scenario). However, a completely different approach to account for LUC emissions has also been proposed, and it was actually included in the earlier version of PAS 2050 (BSI 2008). According to this approach, if the actual LUC emission factor for a certain crop is not known, the highest emission factor for the country of origin in question should be applied. In practice, this would generally mean that transformation from forest to agricultural land is automatically assumed. The idea of this approach is close to the concept of the marginal process, which is generally applied in consequential LCA. For example, according to this approach it can be considered that cultivation of soya is the driving force of the LUC occurring in South America. Therefore, any increase in the demand of soya will automatically lead to increasing LUC (for example clearing of the rainforest), and therefore all related emissions should be allocated to this specific crop. In this paper, this approach is referred as the “Worst case scenario”

2.2. Example problem: GWP of broiler feed

The importance of the selection of the LUCE accounting method is demonstrated in the following by calculating the Global Warming Potential (GWP) for broiler feeds, based on two alternative diets as applied during the whole production cycle. The diets applied here were the standard soy meal-based diet, generally used by broiler industry, and an alternative diet, where part of the soy has been replaced by field peas (up to 30% of the total mass of the feed). Leinonen et al. (2013) gives the more details and the justification of the composition of these diets.

In the calculations carried out for this study, the GHG emissions for soy production were included in the diets using each of the four accounting methods described above, i.e. 1) “Actual LUC”, 2) “Best estimate”, 3) “Top down” and 4) “Worst case” scenarios. The GHG emissions for soy production were calculated as follows. First, following UK import statistics, it was assumed that the soy originates mainly from Brazil (48%) and from Argentina (41%), i.e. from countries where recent LUC related to soy production has occurred. Then, the GHG emissions per land area for different LUC types and countries were specified on the basis of the guidelines given in BSI (2011). After that, the LUCE emissions were allocated to soy beans and subsequently to soy bean meal (and to other crops, depending on the scenario), following each of the above mentioned accounting method. In practice, this was done by using one of the following options: 1) It was assumed that the origin of the soy was known and certified, i.e. it originated from “mature” agricultural land, so no LUCE was allocated to soy (“Actual LUC” scenario). 2) The relative proportion of soy growing in land with a certain land use history, including “new” agricultural land converted from other land use types during the last 20 year period, was estimated on the basis of FAO (2011) statistics, and the weighted average of LUC emissions from these land use types was allocated to soy (“Best estimate” scenario). Leinonen et al. (2013) gives the details of the LUC calculations related to South American soy production. 3) The global LUCE were evenly allocated to all agricultural crops, so the emissions of 1430 kg CO_2 ha^{-1} y^{-1} were applied for all crops used in broiler feed, including soy (“Top-down” scenario). 4) The maximum LUCE, i.e. assuming conversion from tropical rainforest to agricultural land, was applied for all South American soy used in the feed (“Worst case” scenario). Finally, the GHG emissions from feed production, processing and transport were calculated using the method by Williams et al. (2010), and after adding the LUCE, as calculated separately for each scenario, the overall GWP per 1000 kg broiler feed was quantified.

3. Results

The results of the comparison of broiler diets (Table 1) show that the LUCE accounting method has a strong effect on the overall GWP estimate of the broiler feed. For example, in the case of the standard soya diet, the GWP is almost four times as high in the “Worst case” scenario as in the scenario where no LUCE are allocated to soy. However, even a more important consequence can be found when the two alternative diets are compared with each other. The question to be asked here is: does the use of alternative protein sources (in this case field peas) replacing soy reduce the GHG emissions arising from the feed? The answer depends clearly on LUCE accounting method selected. With the “Actual LUC” and the “Top down” methods, no major benefit can be achieved with the use of the alternative pea diet. If the “Best estimate” scenario is used, a relatively high 17% reduction in GWP occurs if the soya diet is replaced by the pea diet. However, with the “Worst case” scenario, a dramatic reduction by 41% can be achieved with the use of the alternative diet. All these different figures can be obtained without any actual physical differences in the systems in consideration; they are only a result of a subjective selection of the method, or traceability of the origin of feed ingredients. Thus the conclusion of agricultural LCA studies that have involved potential LUCE may have been strongly affected by other factors than the actual properties of the system themselves, as discussed below.

Table 1. Global warming potential per 1000 kg broiler feed (kg CO_2e) for either standard soy–based diet, or for the diet where up to 30% peas are used as protein source, using different methods for accounting for LUCE related to soya production.

	Actual LUC[a]	Best estimate	Top down	Worst case
Soy diet	788	1085	1003	3026
30% pea diet	770	902	983	1771
Pea/soy	0.98	0.83	0.98	0.59

[a] Assumes certified sustainable source i.e. no LUCE

The principal differences in the spatial allocation of LUCE of different accounting methods are demonstrated in Figure 1. This figure assumes a theoretical, simplified situation where the global crop production is equally divided between two countries (Country 1 and Country 2). Further, it is assumed that the Country 1 produces one single crop, and half of this crop is produced in an area that has been recently converted from another land use type (e.g. forest). Now in the "Actual LUC" scenario, all LUCE are allocated to the area where they occurred, while in the "Best estimate" scenario only half is allocated to this area, and in the "Top Down" only one fourth. However, in all these cases the global sum of the LUCE is the same, and equal to the actual emissions from the converted area. In contrast, in the "Worst case scenario", although the LUCE of the actually converted area equals the actual emission occurring in that area, the global estimate for the emissions is in this example double compared to its actual amount.

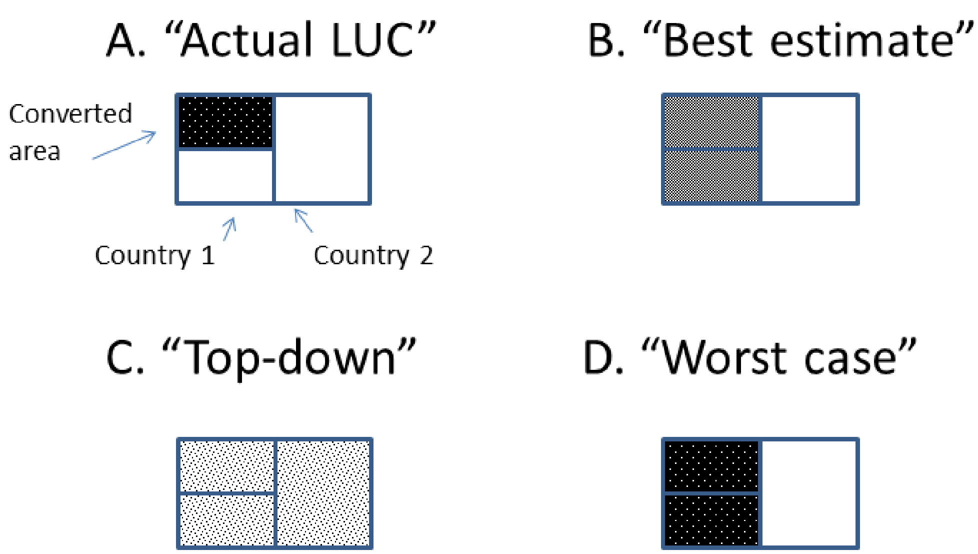

Figure 1. Demonstration of the spatial allocation of LUCE when different accounting methods are used. The darkness of the shading indicates the amount of emissions allocated per unit of land area. In the "Worst case" scenario, the total emissions are estimated to be higher than in the other scenarios, thus indicating double counting.

4. Discussion

When evaluating the alternative LUCE accounting methods against the criteria described above, the key questions are the following: First, does the method follow the principles of attributional LCA, i.e. does it model the systems as they occur or have occurred? Second, does it follow the principle of mass conservation, i.e. accounts for all emissions arising from LUC and allocates them to the products, and at the same time makes sure that no double counting occurs?

In theory, the first approach discussed above, the "Actual LUC" scenario clearly follows all these criteria. It considers a certain, known area of crop production, and as the history of this area is expected to be known, all emissions that have occurred as a result of possible LUC can be quantified, and allocated to the crops grown in this specific area.

It can be considered that the "Best estimate" scenario follows basically the same principles as the "Actual LUC" scenario. The only difference between these two approaches is the spatial scale. While the "Actual LUC" scenario specifies an actual location of certain crop, in the "Best estimate" scenario this area is extended to the whole area of this crop grown in a certain country, or in the extreme case in the whole world. Again, in theory, all the LUC emission within this extended area are accounted for and allocated to the specific crop grown in this same area. Thus, also this approach follows the principles of mass conservation, completeness and consistency.

Although the starting point of the third approach, the "Top-down" scenario, seems different than the first two approaches, as it is based on the concept of including both direct and indirect LUC, it actually can be seen as a further extended version of the "Best estimate" scenario, and it also avoids any double counting. In this approach, all agricultural crops are considered as a uniform group, instead of separating single crops as in the other approaches. The area under consideration is the whole cropland of the world, not the area of any specific crop as in the other approaches. Therefore, it can be concluded that theoretically all these three approaches to LUC accounting are based on the principle of mass conservation, and they fulfil the requirements of ISO 14040, PAS 2050:2011 and the IPCC guidelines. As a result, there is no scientific reason to reject any of these three approaches, and the selection of one of these methods is very much dependent on aim of the study and the availability of the data.

The fourth method, the "Worst case" scenario, is based on the principles of marginal processes, generally applicable in consequential LCA. However, by definition, PAS 2050:2011 is based on the attributional approach, and the same principle is also obvious in the IPCC guidelines. Therefore, it is clear that in any studies following these principles, marginal LUC accounting would not be an acceptable method. The practical consequence when using this approach would be multiple counting of the LUC effect, and therefore considerable overestimation of the GHGs arising from LUC. However, in reality this approach is widely used in literature, also in connection of attributional studies, and this can partly explain the big differences observed in the LUC effect in different studies.

As the results of the broiler feed example above demonstrate, the conclusions of an LCA study on agricultural products, whenever crops with potential LUC are involved, can be solely determined by the selected LUCE accounting method. Probably one of the most significant areas where this problem occurs is the evaluation whether reduction of soya in feeds can reduce the GHG emissions arising from livestock production, and this is where the profound effect of the selection of the methodology has been demonstrated in earlier studies by for example Leinonen et al. (2013) and Meul et al. (2012). In another similar study, Nguyen et al. (2012) quantified the global warming potential of poultry feed, and found a relative moderate potential for reduction of the impact by changing the composition of the feed. One reason for this result is that they used the "Best estimate" approach for LUCE from Brazilian soy (Prudêncio da Silva et al. 2010), based on regional, rather than national averages of production. As a result, the LUCE allocated to soy were relatively low, and therefore removing soy from the diet had only slight effect on the overall GWP. A completely different approach was taken in a recent FAO report, (McLeod et al. 2013), where the global GHG emissions from global pig and poultry production were estimated. In this case the "Worst case" scenario was applied for Brazilian soy, as stated by the authors: "*We thus assume that all incremental soybean area [in Brazil] is gained at the expense of forest area*".

In general, despite its inconsistencies and double counting of the emissions, the "Worst case" approach is apparently still used in agricultural LCA studies. One reason for this is that it was recommended in the earlier version of PAS 2050, the UK specification for assessment of GHG emissions (BSI 2008), before it was replaced by the "Best estimate" approach for its current version (BSI 2011). However, it is interesting that recently some supplementary requirements of PAS 2050 for horticultural products have been published (BSI 2012), and in these requirements further details for LUCE accounting were provided in a way which seem to contradict the general guidelines of PAS2050:2011. In these new requirements, it is stated that: "*To promote the collection of primary data on land use change the principle is adopted that these emissions should not be underestimated. Therefore in deviation of PAS 2050:2011 5.6.2.b, the highest calculated value of the average and weighted average is taken*". This indicates that, despite seemingly conflicting with their Completeness and Consistency principles, the PAS 2050 guidelines are moving back towards the "Worst case" approach.

Although in principle each of the three methods “Actual LUC” scenario, “Best estimate” scenario and the “Top-down” scenario fulfils the requirements of mass conservation, care should be taken when combining these methods, in order to avoid violation of the PAS 2050:2011 consistency principle. For example, it may be possible that in a single LCA study, some of the crop data can be considered to belong to a category “where the country of production is known and the previous land use is known”, while for some other crop production data this cannot be applied. In such a case, either the “Actual LUC” scenario or the “Best estimate” scenario could be applied for separate crops, depending on the traceability of the crop in each dataset. However this combination of the methods is actually against the consistency principle. This problem can be demonstrated with an example where the environmental impacts of soya coming from different sources in Brazil are compared in an LCA study. Assume that one subgroup of this soya is certified so that it is does not cause LUC, and the exact origin of the other subgroup is not known. According to PAS 2050, the “Actual LUC” scenario can be applied to the first subgroup. However, if the “Best estimate” scenario i.e. the LUC effect for “average” Brazilian soya is applied to the other subgroup, this will actually lead to an underestimate of the combined LUC effect of the both groups, as in reality the proportion of the non-LUC related soya in the “unknown” subgroup is smaller than the national average, as part of it is included in the “certified” subgroup.

Another example of the problems of combining the LUCE accounting methods was presented by Meul et al. (2012). In that paper, the authors propose the use of a two-step decision rule to formulate livestock diets with low GHG emissions, i.e. first try to minimize direct LUC (e.g. by using the “Best estimate” approach) and then within this precondition, minimize total land use change risk (i.e. the “Top down” approach) by selecting ingredients with low land use requirements. The problem with this method is that, although technically the LUCE obtained from these two methods are not treated additively, in effect the final conclusions would be based on double counting. Although this principle would effectively eliminate both direct and indirect LUCE from the diets, it would lead to an underestimate the effects of other, potentially significant components of the GWP arising from feed production.

It can thus be concluded that there are three different approaches to account for the LUCE, which can be considered to fulfill the requirements of attributional agricultural LCA, and which differ only in their method of spatial allocation of the emissions. The choice between these methods remains subjective, and is complicated by the fact that despite being justifiable, each of these methods still has its own limitations. Although the “Actual LUC” appears to be able to quantify accurately the LUCE of any crop in consideration, in practice in many cases it could be almost impossible to get specific enough data on the exact origin that crop as this could require site visits to several countries of crop origin. This would be the case especially when the consistency principle is taken into account and double counting is strictly tried to be avoided. This would require that the origin of each crop in any specific study should be precisely tracked, so that a combination of different approaches could be avoided. The problem of traceability can be avoided by using the “Best estimate” approach which has relatively low demands for data if based on national FAO statistics. However, here the question can be raised whether it is justified to make the LUCE country specific, rather than considering the actual global drivers of LUC, or ignoring for example regional differences within a certain country. The idea of the indirect LUC in the “Top down” approach can be justified when considering the land requirement as part of global market of agricultural commodities. In terms of applicability, this approach has an intermediate data need to set up the analysis, but after that it is trivial to implement. However, this approach does not consider any certain crop to be any stronger driver of LUC than any other. A direct consequence of using this method is that higher LUCE are allocated for example to rapeseed oil than to palm oil, just because of the differences of the yield and hence land requirements of these crops.

Due to the limitations discussed above, it seems likely that none of these LUCE accounting approaches, including those which were found technically correct, is likely to become generally accepted “universal” approach to be used in LCA studies. Instead, an attempt should be made to create a new approach that would combine the generally acceptable aspects of the existing methods, while still following all the requirements discussed above. One suggestion for such an approach has been recently presented by Williams et al. (2014), where the “Top down” method is further developed by including the actual drivers of LUC, and still avoiding double counting in spatial allocation of LUCE.

5. Conclusion

Several approaches to account for LUCE of agricultural products appear to fulfil the generally accepted requirements for attributional LCA and avoid double counting. The acceptable methods differ in their spatial allocation of the emissions, and the selection between them remains subjective. However, it should be possible to combine the benefits of several methods to achieve only one generally accepted approach to be used universally in agricultural LCA. A modified "Top down" method, weighted for different crops based on their roles as actual drivers of LUC, might be one option.

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Modelling of nitrogen releases in life cycle assessment of crop production

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ABSTRACT
The modelling of reactive nitrogen (Nr) releases poses a big challenge to the life cycle assessment (LCA) of crop production. This study aims at providing an overview of potential aspects that need to be taken into account for improved modelling of Nr releases in the LCA of crop production. The definition of a crop product system and considerations of the crop, the soil, and the spatial scale were revisited. The major pathways of releases of nitrate, nitrous oxide, and ammonia were distinguished. Empirical emission-factor-based methods and process-based models for the development of unit process dataset of on-site crop production were summarised and compared. The importance of modelling fates of Nr releases was discussed. At the end, several conclusions were drawn.

Keywords: nitrate, nitrous oxide, ammonia, release, crop production

1. Introduction

Crops are plants that can be grown and harvested extensively for profit or subsistence (Encyclopædia Britannica 2014). Crop production, whist contributing to ensuring ample food and feed supply and enabling adequate farmer income, has induced numerous detrimental impacts on the quality of air, water, and soil, terrestrial ecosystems and biodiversity, greenhouse gas balance, etc. (Tilman 1999; National Research Council 2010). Crop production is the single largest cause of human alteration of the global nitrogen cycle (Liu et al. 2010). Life cycle assessment (LCA) has been identified as a valuable tool for the evaluation of potential environmental impacts of farming systems (van der Werf and Petit 2002) and applied increasingly to crop production as well as animal production (Brentrup et al. 2001; Haas et al. 2001; Bessou et al. 2013; Perrin et al. 2014).

However, originally developed for industrial production, the application of LCA to crop production faces challenges, which are determined by the features of crop production that are more complicated than industrial production. A main complicating feature is that crop production uses the soil. The dynamics of crop nutrients such as nitrogen, phosphorus, and potassium as controlled by biophysical soil processes need careful consideration (Harris and Narayanaswamy 2009). The development of models to estimate on-site pollutant emissions of crop production has been identified as a topic that leads the methodological development in LCA (van der Werf et al. 2014).

Amongst crop nutrients, nitrogen is of crucial importance, as it is a major determinant of crop growth and yield (Goulding et al. 2008). The large increases in reactive nitrogen substances (Nr) as fertilisers used in crop production, observed after the Second World War in many parts of the world, have triggered a cascade process generating Nr releases to air, soil, and water at each stage of the cascade (Galloway et al. 2003). Nr have been systematically catalogued in life cycle inventory (LCI) results and assessed against several important potential environmental impacts in life cycle impact assessment (LCIA).

The aim of this study is to provide an overview of potential aspects that need to be considered for improved modelling of Nr releases in the LCA of crop production.

2. Goal and scope definition

2.1. The crop product system

In LCA studies the boundary should be delineated clearly between the product system and the ecosphere. In the LCA of crop production, a crop product system encompasses both the upstream processes, i.e., the extraction of natural resources and the supply chain of various intermediate products such as fertilisers, pesticides, machinery, infrastructure, energy, etc., and the on-site crop production. The crop delivered at the farm gate is used for several possible functions such as food, livestock fodder, biofuel stock, clothing, medicine, etc. (A detailed classification of crops and crop functions can be found in the FAO World Programme for the Census of Agriculture (2010)).

The analysis of on-site crop production involves modelling consideration in three dimensions, vertically about the crop and the soil and horizontally about the spatial scale. The harvested portion of the crop and the remaining non-harvested portion are regarded as part of the product system and part of the ecosphere, respectively. Soil can either be included in the ecosphere based on the hypothesis that damage to the soil should be regarded as an environmental impact so as to distinguish systems differing in their impacts on soil quality, as suggested by Wegener Sleeswijk et al. (1996), or be regarded as an integral part of the product system, as proposed by Audsley et al. (1997). This study follows Audsley et al. (1997) to take into account the crop and the soil with a depth right down to the water table, which is location specific and changes along with seasons, in delineating the crop product system. On-site crop production can be modelled and analysed on a range of spatial scales, from the field through the farm to the farming region or even broader scales (e.g., a nation). The different spatial scales may include different compartments of the ecosphere and thus imply different fates of Nr releases in the assessment.

2.2. Major Nr under consideration

A non-exhaustive list of various forms of Nr under consideration in LCA studies is provided in Table 1.1. The three most important Nr releases in the LCA of crop production are nitrate (NO_3^-), nitrous oxide (N_2O, a.k.a. dinitrogen oxide), and ammonia (NH_3), which contribute to the impact categories of terrestrial and aquatic eutrophication, climate change, and acidification.

Table 1. Overview of reactive nitrogen substances and their contributed environmental impacts in LCA

Nr Substance	formula	Impact category
Nitrate	NO_3^-	Eutrophication
Nitrite	NO_2^-	Eutrophication
Ammonia	NH_3	Acidification and eutrophication
ammonium	NH_4^+	Acidification and eutrophication
Nitrous oxide	N_2O	Climate change and ozone depletion
Nitrogen oxides	NO_X	Acidification, eutrophication and photochemical ozone formation

2.2. Pathways of Nr releases

Based on the aforementioned definition of a crop product system, two types of Nr releases can be distinguished: upstream Nr releases and on-site Nr releases, which are elementary flows from the crop product system to the ecosphere. Upstream releases are associated typically with inputs to the on-site production, for instance, the release of NOx due to fuel use. Their data are usually obtained from existing databases such as ecoinvent (Swiss Centre of Life Cycle Inventories 2010). Whilst increased Nr mineralisation/nitrification can be a significant source, on-site releases are mainly due to the applied nitrogen fertilisers, which are transformed and transferred in the soil and taken up by crops that are modelled as part of the product system. Crop residues can be a major contributor to the release of N_2O (Baggs et al. 2000). Most on-site Nr releases are attributed to a range of transformations including ammonia volatilisation, nitrification and denitrification, and to the transfer in terms of nitrate runoff and leaching and organic Nr runoff, as shown in Figure 1 (Parnaudeau et al. 2012) (Note that processes of exportation and fixation in Figure 1 are actually outside of the crop product system).

Outside of the crop product system, upstream and on-site Nr releases experience various transformations and affect the environmental quality and human health. These transformations correspond to the cause-effect chain between releases and environmental impacts and are the basis for modelling nitrogen-related impacts in LCIA. Table 2 summarises the major pathways that are within and outside of the crop product system.

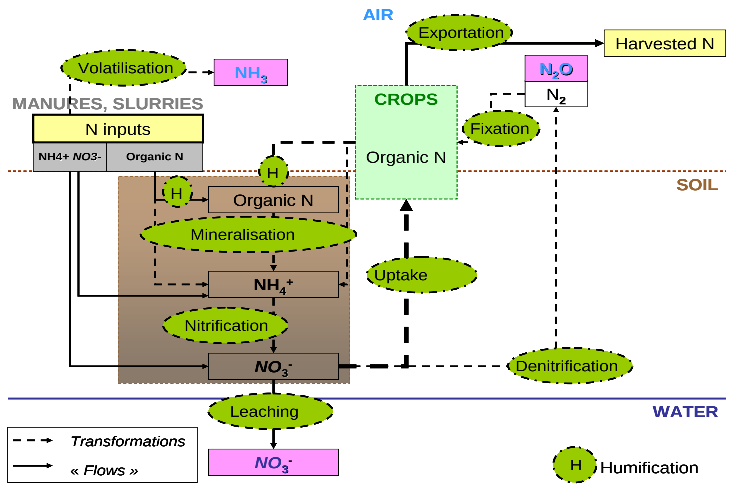

Figure 1. Nr releases from the on-site crop production (Parnaudeau et al. 2012)

Table 2. Overview of pathways of Nr releases related to crop production

Nr	Within crop product system		Outside crop product system
	Upstream release	On-site release	Transformation
NO_3^-	Discharge	Leaching and runoff	Denitrification, ammonification
N_2O	Emission	Denitrification of NO_3^- in soil, crop residues	Denitrification, nitrification
NH_3	Emission	Volatilisation	Nitrification

3. LCI related to Nr releases

Nr release rates can vary greatly depending on farmer practices, weather conditions, soil characteristics, landscape characteristics, hydrological processes, etc. (IFA/FAO 2001). On-site measurements of the Nr releases are costly and time-consuming, and in any case display great variations, because they only reflect a snapshot of the specific conditions of the on-site crop production at the time of measurement (Brentrup et al. 2000). In practice, the unit process dataset of on-site crop production is seldom built based on on-site measurements but rather created by linking raw data from various sources with pertinent mathematical relationships. Two types of mathematical relationships are distinguished, i.e., empirical emission-factor-based methods and process-based models.

3.1. Empirical emission-factor-based methods

An emission factor (EF) is defined as the average emission rate of a given (gaseous) pollutant for a given source, relative to units of activity. Emission factors are traditionally used to quantify air, water and soil pollutants at relatively high aggregation levels (e.g., the national scale), rather than at the individual source of emissions. In most cases, these factors are simply averages of all available data of acceptable quality. For instance, a default EF of 0.01 (range: 0.3–3%) was suggested by the Intergovernmental Panel for Climate Change (IPCC, 2006) to calculate the N_2O emission due to fertiliser application to managed soil at the national scale.

Bessou et al. (2013) and Perrin et al. (2014) reviewed 155 publications on perennial crop production and 17 publications on vegetable crop production, respectively. Three common references were identified as the general guidelines in estimating “field emissions” (i.e., on-site Nr releases), viz. Audsley et al. (1997), Nemecek and Kägi (2007), and Brentrup et al. (2000), as summarised in Table 3. In addition, the 4th report of the Intergovernmental Panel for Climate Change (IPCC, 2006) was also identified as an important guideline for perennial crops by Bessou et al. (2013).

Table 3. Empirical emission-factor-based methods for estimating on-site Nr releases of crop product systems (following Bessou et al. (2013) and Perrin et al. (2014))

Nr	Audsley et al. (1997)	Brentrup et al. (2000)	Nemecek and Kägi (2007)
NO_3^-	n.a. [a]	$N\text{-}NO_{3}^{-}{}_{,soil}$	n.a. [a]
N_2O	EF% of $N_{fertiliser}$	EF=1.25% $N_{fertiliser}$– NH_{3loss}	EF=1.25% N_{soil}
$NH_{3,fertiliser}$	EF% of $N_{fertiliser}$	EF% of $N_{fertiliser}$	EF% of $N_{fertiliser}$
$NH_{3,manure}$	EF=50% of $N\text{-}NH_{4,manure}$	$N\text{-}NH_{4,manure}$	$N\text{-}NH_{4,manure}$

[a] n.a.: not available in English.

3.2. Process-based models

Process-based models try to represent physical/biological/hydrological processes observed in the reality (Korzukhin et al. 1996). They are spatialised (semi-distributed or distributed) models, which can take into account different scales of representation of crop production (field, farm, farming region, etc.) and the interactions between the subsystems of the crop product system. Typically, for the estimation of releases from crop production such models contain representations of surface runoff, subsurface flow, evapotranspiration, and channel flow. Process-based models are increasingly recognised as alternatives to empirical emission-factor-based methods. We selected several process-based models through a search for peer reviewed English publications in the ISI Web of Knowledge (accessed on 9 April 2014), which is summarised in Table 4.

Table 4. Selected process-based models for estimating Nr releases from crop production

Process-based model [a]	Reference	Simulated Nr	Scale
CERES-EGC	Gabrielle et al. (2006), Bessou et al. (2013)	N_2O	Farming region
DNDC	Li (2000) Deng et al. (2011)	NO_3^-, N_2O, NH_3	Farming region, nation
DayCent	Parton et al. (1998), Del Grosso et al. 2005)	N_2O, NO_x	Farming region
DNMT	Liu et al. (2005)	NO_3^-	Farming region
SWAT	Arnold et al. (1998), Gramig et al. (2013)	NO_3^-	Farming region
FASSET	Jacobsen et al. (1998), Chatskikh et al. (2005)	NO_3^- and NH_3	Field, farm
HERMES	Kersebaum (2007)	NO_3^- and NH_3	Field
TNT2	(Beaujouan et al. 2002)	NO_3^-, NH_3	Farming region
INCA	(Wade et al. 1999)	NO_3^-, NH_4^+	Farming region

[a] CERES-EGC: Crop Environmental REsources Synthesis; DNDC: DeNitrification-DeComposition; DayCent: Daily Century; DNMT: Diffuse Nitrate Modelling Tool; SWAT: Soil and Water Assessment Tool; FASSET: An Integrated Economic and Environmental Farm Simulation Model; TNT2: Topography-based Nitrogen Transfer and Transformation; INCA: Integrated Nitrogen in Catchments model

As two types of mathematical relationships to create unit process datasets from raw data, empirical emission-factor-based methods and process-based models differs for several characteristics, as indicated in Table 5. Process-based model can simulate the Nr dynamics related to the crop product system by parameterising various influencing mechanisms. Data representativeness and model uncertainty depend on the data availability, which will often determine which approach is used. Whenever sufficient data are available, process-based models are recommended. However, special attention should be paid to ensure the consistent spatial scale of stand-alone LCA studies of crop production and those coupled with process-based models.

Table 5. Comparison between empirical emission-factor-based methods and process-based models in LCA of crop production.

Characteristic	Empirical emission-factor based methods	process-based models
Mathematical relationship	Stochastic	Deterministic
Data requirements	lower	Higher
Data representativeness	lower	Higher
Primary error source	Extrapolation	Unknown parameters
Model uncertainty	Lower	Higher

4. LCIA related to Nr releases

The fate of an on-site Nr release consists of its transfer, transformation, and accumulation or dilution in a compartment of the ecosphere (Basset-Mens et al. 2006). The fate of a specific release plays an important role in the cause-effect chain linking its release to its potential environmental impacts and needs to be carefully modelled. Considering the aforementioned definition of the crop product system, such a compartment can be the atmosphere, the hydrosphere, or part of the pedosphere that is not included in the crop product system. The formation of N_2O outside of the crop product system (cf. Table 2) indicates that the fate factors of leached NO_3^- and volatilised NH_3 from the on-site crop production are both intrinsically inferior to 1 in the hydrosphere and the atmosphere, respectively. Fates of an Nr releases are usually ignored by assuming them equal to 1 excepted in rare studies such as Basset-Mens et al. (2006). Basset-Mens et al. (2006) looked into the fate factors of NO_3^- releases in three different catchments in west France, based on the fate factor for NO_3^- which was defined as the ratio of the NO_3^- quantity at the outlet of a catchment over the NO_3^- discharged from the catchment's soils. It can be a valuable reference for future studies on simulating the fate, for instance, of NH_3.

5. Conclusion

The following aspects need to be taken into account for improved modelling of Nr releases in the LCA of crop production:

- The crop product system consists of upstream processes and the on-site crop production. The on-site crop production includes the harvested portion of the crop and the soil with a changing depth down to the water table
- Nitrate, nitrous oxide, and ammonia are three important reactive nitrogen substances under consideration. They should be distinguished within the crop product system and between the crop product system and the ecosphere.
- Empirical emission-factor-based methods and process-based models are typical approaches for the development of unit process dataset for on-site crop production. Stand-alone LCA studies of crop production and those coupled with process-based models should be based on the consistent spatial scale.
- Fates of reactive nitrogen releases in the ecosphere should be explicitly modelled in the life cycle impact assessment phase of the LCA of crop production.

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Feed and Food Databases in LCA – An example of implementation

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ABSTRACT

New LCA databases in the feed and food sector have recently been published or are announced to be forthcoming. The objective of this publication is to present key aspects of implementation of one of these databases, the "GaBi feed and food database" (to be released in autumn 2014). Firstly, the GaBi LCA agrarian plant model, which has been continuously developed over the past decade and is embedded in a methodological and scientific framework, serves as a basis for modelling feed and food products. The model is based on a synthesis of relevant scientific and methodological approaches which are described in this publication. The specific approaches chosen to address the agricultural LCA modelling challenges of nitrogen cycle, reference system, CO_2 uptake and storage in biomass, land use, land use change, irrigation and agricultural chemicals are described. Secondly, the approach to allow for a scientifically sound, feasible, consistent and applicable scaling of dataset generation for use in industry is explained in principle. Parameters in the model have been identified, analyzed and grouped and are determined via sequences of data collection. Thirdly, the modelling approach includes management, update and maintenance procedures to enable the data to follow the natural, agricultural, political and technical dynamics, prices and supply chain key parameters in time. The management and maintenance concept includes a professional systematic review process with yearly update schedules.

Keywords: Feed and Food, LCA, agrarian plant model, database development, GaBi

1. Introduction

The feed and food industry is considered to have a major contribution on the overall anthropogenic impact on the environment (FAO 2006, IPCC 2014a, IPCC 2014b). Several reports on strategies to abate this impact propose to focus efforts on energy efficiency, emissions efficiency and on a change and reduction of demand (IPPC 2014a). The latter implies action and interaction of consumers, governments and industry, for example to decrease the amount of lost, wasted, unused food (FAO 2006, FAO 2013). Producers of feed and food are also responsible for identifying and applying suitable technical solutions for a more sustainable and efficient use of resources. The possible benefits for companies are manifold, including potential cost savings and proactive mitigation of potential harm on their business by changing the environment and the environmental resources on which they depend (Callieri et al. 2008).

Life cycle assessment (LCA) can be used to achieve these objectives: LCA is a tool used by companies to benchmark and implement sustainability measures over an improvement processes into their daily operations and business. Many companies that successfully implemented sustainability measures conclude that "you can only improve what you can measure" and that "only what gets measured gets managed" e.g. (Warsen 2013). The relevance of implementing applicable approaches in the agri-food practice is underlined by the estimate that around 25% of all LCA studies are related to agriculture (Blonk 2014a).

Besides ethical and economic reasons for the sector to be actively engaged in sustainability, in particular LCA, several drivers increase the importance of LCA in the agri-food sector: policy developments, such as the green market initiative of the European Commission and its Product Environmental Footprint (PEF) pilots, sector specific initiatives, such as the ENVIFOOD protocol (ENVIFOOD 2013), FAO LEAP (FAO LEAP 2014), Feed LCA guidelines (IFIF 2014) and others offer new possibilities for the topic and for companies.

Different stakeholders, including companies, universities and external data providers, are working independently on data solutions to respond to an increased data demand. Many databases were recently published or are forthcoming: "Agri-footprint®", "AGRIBALYSE®", "World food LCA database", "Feed and food database" (see references for details on the databases). The main objective of this publication is to describe 3 key aspects of implementing the "Feed and food database", which is released in autumn 2014.

Many feed and food products are based on raw materials derived from agricultural plant systems. Different processing steps, agricultural management, farming and plant growing, handling, packaging, transport are required before a final product is ready for use (and, at end of life, specific disposal scenarios may need to be considered). The focus of this publication is on the plant system part that forms the data backbone of the “Feed and food database”. This is the major aspect covered in this publication.

Agrarian plant systems belong to complex production systems within LCA due to their dependence on environmental conditions that are variable in time (e.g. within a year, from year to year) and in space (e.g. varies by country, region, local site conditions). Also the correlation between inputs (of fertilizers, pesticides, agricultural engineering, etc.) to outputs (of harvested crop, gaseous field emissions, leachate, etc.) is complex and often non-linear in nature. The following factors contribute to the complexity of agricultural modeling:

- The variety of different locations,
- Small scale soil variability within and between locations,
- The large number of farms,
- The variety of agricultural practices (e.g. conventional vs. organic) and equipments,
- Technically, there is no well-defined border with the environment (´open system`),
- Complex and indirect dependence of the output (harvest, emissions) from the input (fertilizers, location conditions etc.),
- Variable weather conditions within and between different years,
- Different cultivation periods (e.g. annual, perennial, plantations),
- Quality and properties of products,
- Multi-output systems (product(s) and by-products) and allocation diversity,
- Various and big amount of Functional Units (FU),
- Farming intensity (e.g. extensive vs. intensive),
- Water availability and importance,
- Variable pest populations (insects, weeds, disease pathogens, etc.),
- Different cropping systems (e.g. monoculture, polyculture), and
- Political conditions influencing cultivation.

In addition to the appropriate consideration of these technical modelling aspects, customers of LCA datasets and databases want datasets to be highly reliable, to be easily understandable and marketable and to be delivered at a reasonable cost and time (Deimling et al. 2008). The transparency of the underlying model, of the raw data sources, of the applied QA procedures and of the external review processes are also of high importance to ensure the required professional credibility.

2. Methods

The GaBi LCA agrarian plant model is the basis for all standard agrarian datasets that are provided in GaBi databases. The following requirements were identified by the authors during their experience over the last 12 years as being crucial for an integrated approach to develop new agri-food datasets:

1) Scope: agrarian models (in this publication the focus is on terrestrial agrarian plant systems)
2) Scale: agri-food data development and supply, based on clearly defined processes and quality assurance
3) Management & maintenance: frequent updates, reviews and improvements of data and databases

These requirements are explained in more detail in the following sections. The precondition for these 3 requirements is a framework, which includes a collaborative review of suitable existing scientific methods, databases and inclusion of identified aspects (this requirement is not covered in this publication).

2.1. Scope

Due to the inherent complex characteristics of an agricultural system, a relatively extensive and comprehensive, nonlinear, computing model for agrarian plant and plantation processes was developed and implemented in the GaBi software (Deimling et al. 2008). The model includes a multitude of input data, emission factors and parameters and its proper application of the model necessarily requires both agricultural expertise and LCA experience.

Each processing step has been broken down into separate production modules such as soil cultivation and preparation, weed control, pest control, fertilization including organic fertilizers, harvest main product, by-product etc., and others (i.e. irrigation). For each of these modules data on the inputs and outputs (i.e. fertilizers, pesticides, fuel for machinery, etc.) have to be collected and calculated.

As shown in Figure 1, key characteristics of the model are different implicit mapped compartments (in particular: field operations, general agricultural process, land use change, clearing process, agrochemical processes, reference system land use, land use, allocation, carbon correction, soil leaching, irrigation process). This structure creates a comprehensive, flexible model that is highly parameterized and includes many background processes. As such, it is applicable for assessing any agrarian and plantation product of the world. Parameters can be fine-tuned to a high degree while the operating effort is reasonable. This allows inclusion of e.g. effects of crop rotation and periods with limited plant growth. The advantage and prerequisite is the delivery of consistent and exact results (Deimling et al 2008).

Besides life cycle inventories, different life cycle impacts assessment methods for environmental indicators and impact categories can be applied.

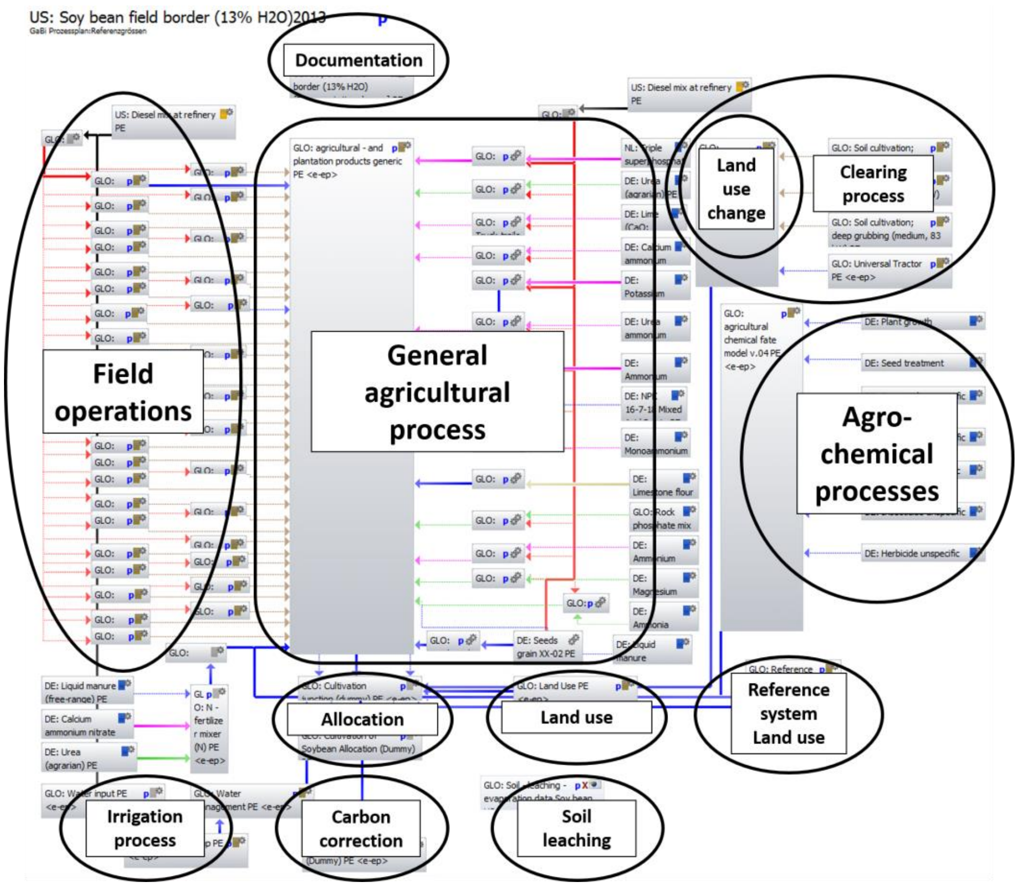

Figure 1. Screenshot of the GaBi LCA agrarian plant model with illustration of different modelling modules.

The system boundaries of the GaBi LCA agrarian plant model encompass agricultural production as well as post-harvest processes and transportation, and are shown in Figure 2. Some aspects were excluded from the GaBi LCA agrarian plant model as their relevance to the environmental profiles was found to be low, these include:

- In case of human labor, social issues are outside the scope
- Construction of capital equipment and maintenance of support equipment, and
- Packaging materials for seeds, fertilizer, pesticides etc.

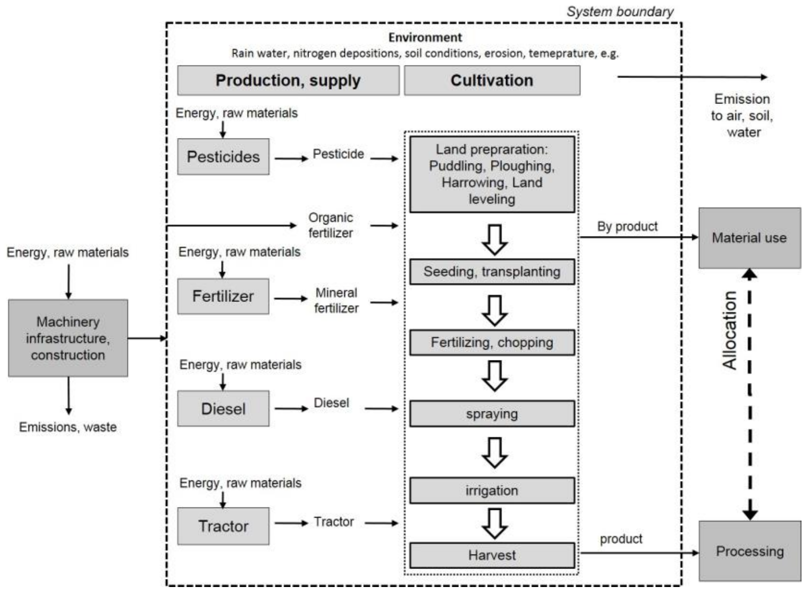

Figure 2. System boundaries of the GaBi LCA agrarian plant model.

To illustrate the background of the GaBi LCA agrarian plant model, seven key functionalities of the model are further explained below.

Nitrogen cycle (as part of the modeling module "general agricultural process"): Nitrogen plays a fundamental role for agricultural productivity and is also a major driver for the environmental performance of an agricultural production system (Eickhout et al. 2006). For these reasons it is essential to evaluate all relevant nitrogen flows within, to and from the agricultural system. GaBi LCA agrarian plant model accounts for the nitrogen cycle that occurs in agricultural systems. The model ensures that nitrogen emissions are consistent for all cultivated species. Specifically the model includes emissions of nitrate ($NO3^-$) in water and nitrous oxide (N_2O), nitrogen oxide (NO) and ammonia (NH_3) into air. The different N-based emissions are calculated as follows:

- NH_3 emissions to air from mineral and organic fertilizers are adapted from the model of rentrup and modeled specifically for the cropping system dependent on the fertilizer-NH_4^+ content, the soil-pH, rainfall and temperature. The following emission factors are used by default and adjusted in case more specific information is available. For ammonium nitrate, calcium ammonium nitrate, monoammonium phosphate, diammonium phosphate 2% of fertilizer N input, for ammonium sulphate 10%, for urea ammonium nitrate 8% and for urea 15% are emitted. These values are identical with data from Döhler et al. 2002 (with exception NH_3-N).
- N_2 is the final product of denitrification. Denitrification is a process of microbial nitrate reduction that ultimately produces molecular nitrogen through a series of intermediate gaseous nitrogen oxide products. N_2 emissions are assumed to be 9% of the N-fertilizer input based on a literature review made by Van Cleemput 1998. N_2 emissons are also taken into consideration to determine the nitrate leaching potential.
- NO is an intermediate product produced in microbial denitrification. NO emissions are calculated from the reference system after N-input from air plus 0.43% of the N-fertilizer input specific for the cultivation system as NO according to Bouwman et al. 2001.
- N_2O is an intermediate product produced in microbial denitrification. According to IPCC 2006, N_2O emissions were calculated as 1% of all nitrogen available including nitrogen applied with fertilizers, atmospheric deposition, microbial nitrogen fixation, nitrogen available from previous crop cultivation and indirect emissions.

- NO_3^- emission to groundwater is calculated based on available nitrogen (N not lost in gaseous form or taken up by the plant, stored in litter, storage in soil, etc.). Depending on the leaching water quantity and soil type, a fraction of this available nitrogen is calculated to be leached nitrate. When available N is calculated to be negative, (for instance due to a higher extraction than fertilizer input) a minimum N-loss factor is applied on the applied fertilizer and N_{min} quantity.
- N_{org} and NO_3^- emissions to water occur due to erosive surface run-off. It is very difficult to generalize erosion rates and deposition rates, as they are highly dependent on regional conditions such as climate, relief, soiltype, crop cultivated and vegetation. Soil erosion rates are estimated based on USDA 2003 and Wurbs and Steiniger 2011. It is assumed that 10% of the eroded soil accesses the waters, based on an evaluation of different literature sources Fuchs and Schwarz 2007, Hillebrand et al. 2005, Helbig et al. 2009, Nearing et al. 2005, while the rest accumulates to colluviums on other surfaces and is assumed irrelevant in the life cycle assessment.

Compared to a pure N-balance model, this approach allows the illustration of N-losses in case of very low N-fertilization (e.g. N-deficit in rubber-tree plantations). In the case of high N-fertilization (e.g. intensive farming applications), the models correspond with the total N-balance approach. Emissions from erosion, the reference system, and nutrient transfers within crop rotations are also modeled consistently.

Reference system (modelling module “Reference System Land Use”): Reinhardt 1998 (amongst others) illustrated the importance of usage of a reference systems in agricultural systems. The reference system is an inverse process used to assess the behavior of land that is not used agriculturally or influenced anthropogenically. In particular losses of nitrate to groundwater and emission of gaseous nitrogen compounds that result from nitrogen deposition onto this land are considered. This takes place in both the main cropping system as well as on land not under cultivation. Therefore not all occurring emissions can be assigned to the crop as they also occur on non-cultivated land, e.g. if this is fallow or a nature reserve. Here it is assumed that the nitrogen balance is neutral for the reference system, as any entry of nitrogen with rainfall is re-emitted from the systems in various forms into ground water and air.

In addition to the emissions of nitrogen compounds, the soil erosion is mapped including the associated conditional entries of organic carbon contained in the soil and some heavy metals in surface waters. The same principle is applied that this erosion occurs to a lesser extent also in non-utilized natural systems and therefore cannot be assigned completely to the main crop.

CO_2 uptake and storage in biomass (part of modelling module “general agricultural process”; if allocations are applied, modelling compartment “carbon correction”): The product bound CO_2 has to be accounted for directly as 100 % on the input side (flow: carbon dioxide resources) comparable to CO_2 emissions into air on the output side (flow: carbon dioxide biotic). The CO_2 quantities from renewables emitted during later stages in the life cycle (e.g. burning, composting etc.) have to be accounted as emissions to air (ILCD 2010). For fast moving consumer goods, this means that over the life cycle all bound CO_2 is released at a later stage. Carbon emissions (besides CO_2 e.g. CH_4 and CO) during biomass production, its conversion and its end of life are also considered.

In case of allocation, the carbon uptake of the product (after allocation) is corrected to the carbon stored in the product via an adjustment of the carbon dioxide resources flow.

Land Use (modeling module “land use”): Further inventory data on land use is provided site specifically for the foreground system using the LANCA method (Beck et al. 2010). In order to include land use issues, the impacts on ecosystem services are considered, especially for the indicators erosion resistance, mechanical filtration, physicochemical filtration, groundwater replenishment and biotic production.

Irrigation (modeling module “irrigation process”): water use is modeled based on the calculations of Pfister et al. 2011. A generic water model allows the selection of different plant water requirements and irrigation regimes depending on the specific regional conditions (e.g. precipitation, (fert)irrigation demand and irrigation technique). For details please refer to Pfister et al. 2011 and also Thylmann et al. 2012.

Agricultural chemicals (modeling module "agrochemical processes"): eco-toxicity is an indicator that is becoming increasingly important in LCA. Therefore a close look at pesticide use and its environmental impact is necessary when performing a LCA of agricultural products. Both the production and use of pesticides contribute to their environmental burden. The life cycle inventories of pesticide production are based on literature data as primary data is only very rarely available. The LCIs cover all input and output flows relating to pesticide production. All pesticide LCIs are based on representative LCIs (background datasets) that are valid for all active ingredients within the same effect category (herbicide, fungicide, etc.). All available data on energy use in pesticide production is discussed in Audsley et al. 2009. A Pest LCI approach was integrated into the model based on the approach described by Dijkman et al. 2012. The Pest LCI model provides emission factors for active ingredients (AI) to water (ground and surface) and air. About 125 scenarios with different parameter settings for pesticide AI, crop and location were calculated and are available as parameters in the GaBi LCA agrarian plant model. In case of more specific data availability these parameters are specifically adjusted.

Land use change (modeling module "land use change")**:** Emissions from direct land use change were calculated with the direct land use change assessment tool (Blonk 2014b) for the approach "weighted average" (as mandatorily required by the Envifood protocol and in line with WRI/WBCSD GHG protocol requirements) based on the approach from PAS 2050-1:2012 and WRI/WBCSD GHG protocol. This approach is crop-specific: The impacts from land use change are allocated to all crops, which increased in area harvested in a specific country, dependent on their respective share of area increase. According to all 3 standards, these emissions are distributed over a time period of 20 years. The tool works with statistical data from FAOSTAT for crop yields, harvested area of crops and area of forest and grassland, from FAO's global forest resource assessment for carbon stocks (in case former land use is unknown) (FAO 2010), from EC JRC world map of climate types and world map of soil types (EC 2013a) and from IPCC for above ground mass carbon stock (if land use change is known), values of soil organic carbon stock and stock change factors (IPCC 2006a). Changes in soil organic carbon stock are taken into account with that methodology. The emissions are reported separately with a flow "carbon dioxide from land use change" as required by certain guidelines (e.g. ISO/TS 14067).

Indirect land use change (please see EC 2010 for definition) is currently not considered as there are no international accounting standards available and while LCA is based on physical flows iLUC is based on market predictions (Finkbeiner 2014).

2.2. Scale

Deimling et al. 2008 already concluded that some challenges remain as agricultural systems are complicated, so the model developed to assess them is also complicated and both data- and resource-intensive to use. Questions regarding the scalability of LCA in the feed and food industry were raised, especially with focus on retailers, who may stock thousands of food items but who do not manufacture these themselves. A need for "a streamlined data collection processes or central resource where the data requirements of the model can be easily accessed" was identified (Deimling et al. 2008).

To ensure the high feasibility and applicability of the GaBi LCA agrarian plant model the significance of parameters was identified by the authors of this study, based on general LCA experience and specific experience with the GaBi LCA agrarian plant model and other literature (e.g. Nemecek and Schnetzer 2011). According to their importance, the parameters were prioritized by grouping them into three different groups, which are shown in Figure 3.

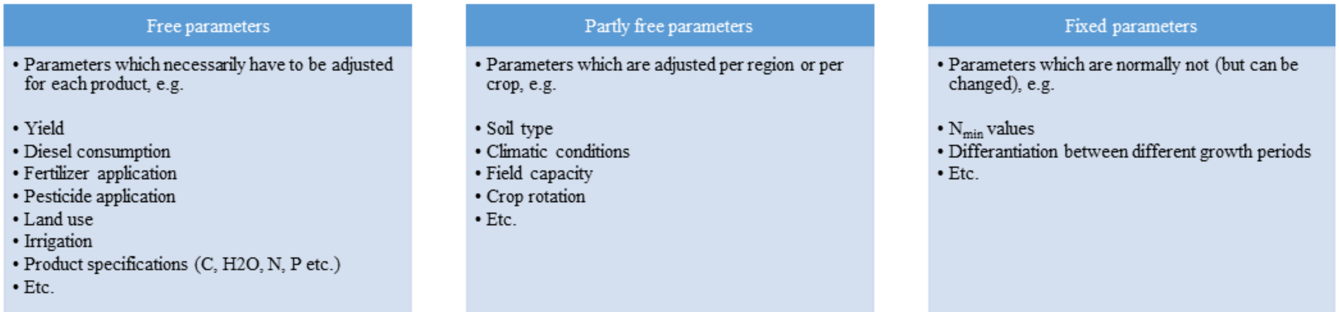

Figure 3. Display of three different parameter groups for agrarian plant modelling with the GaBi LCA agrarian plant model.

Sequences of data collection for parameter determination were defined, including flow diagrams for all "free" parameters. These flow diagrams and the parameter list will be made publically available in the documentation of the "feed and food database", which is released in autumn 2014. The data collection process always starts with specific primary data - if available - before moving to more generic data. The most specific data are used preferentially. A framework for determination of sources is provided to the LCA practitioners in charge of data development. The most generic solution is an expert judgment of discrete, quantified steps which lie in a predefined and suitable bandwidth of values. This estimation of parameters is the last resort option if no better data can be obtained elsewhere.

Full transparency of the approach is essential for all aspects relating to "management & maintenance" as described in the next section.

2.3. Management & Maintenance

This chapter discusses how to convert agrarian plant LCA models into broadly usable data, while maintaining its scientifc soundness and technical quality over time.

Feed and food data are the base and fuel for related LCA results. However in LCA different "planets" of users exist (Klöpffer and Heinrich 2001). These have different backgrounds but still talk about the "same" tasks and concepts. As a result, professional stakeholders have sought to develop approaches that can reduce misunderstandings and improve credibility (e.g. Rebitzer 2001). Even though different stakeholder backgrounds and some miscommunication still remains a decade later (Baitz et al. 2012), it is time to move another step forward.

Stakeholders all aim to support LCA, but each one has his own, often a different interpretation of what this means. However for application in agroindustry it is important to consistently base their work on reliable and risk mitigating professional data solutions. Reliability, consistency and conformity of data are, besides scientific soundness, key for success for professional use.

A professional environment necessarily includes reviews, verification, continuous improvement and yearly updates. To move from "scientific sound models" towards "scientific sound models that are constantly reviewed, maintained and updated" certain maintenance and quality assurance processes must be implemented. Consistency, continuity and reliability are core features of technology development, provision and maintenance for LCA software and database providers. The challenge is to manage the different inputs of stakeholders in a way that valuable data can be released and published in a consistent framework while immature or fuzzy data are filtered out. Data provision and use should be understood by any stakeholder as part of the "normal" management cycle: Plan-Do-Check-Act (or, to translate into the LCA data world, maybe: Plan—Implement—Review–Maintain).

Figure 4 explains the external review and auditing process which is implemented in PE INTERNATIONAL and also true for the feed and food database. The figure shows the development of databases over time (for the time period 2012-2014; whereas the procedure continues after 2014). Jira is used as program for quality assurance to allow users and employees to report issues, improvements on the LCA software and LCA database content. The treatment of these reports issues is documented and continuously reviewed by an external auditing company (DEKRA). Spot checks were performed by ENEA and Ciemat on behalf of the European Commission

(EC 2013b, ENEA 2014). Databases are updated once per year – inconsistencies are corrected immediately. Figure 5 shows the main review and audit aspects, displayed in a circle, which underpins the interaction and sequence of the different aspects.

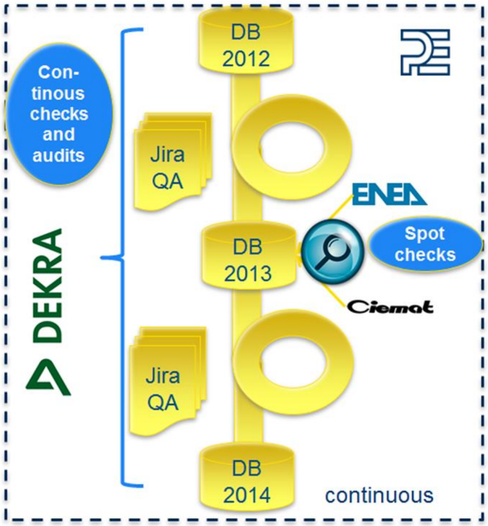

Figure 4. External Review and Auditing in a periodic continuous improvement setting of LCA databases (DB).

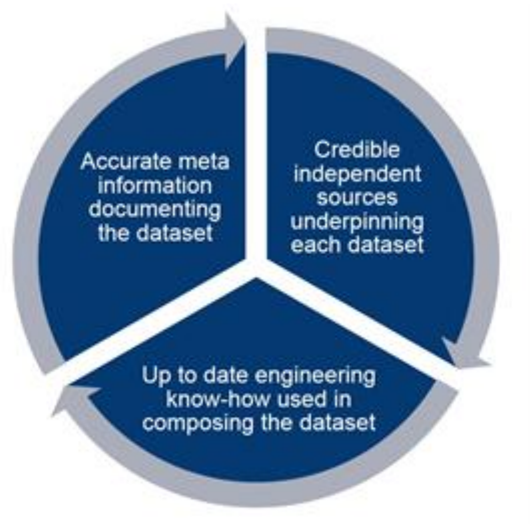

Figure 5. Main review and audit aspects.

"Demand" is the core driver when planning data provision. New technologies, regulations, standards or new market regions are decisive. The core aspects concerning implementation of data are: overall relevancy, accuracy, methodological consistency and technical adequacy. Data implementation needs some LCI experts and many engineering and agri-food experts to generate adequate data results. Concerning maintenance of data, the frequency, the possibility of auto-updates of own-developed user systems and the proactive update or fade-out of older data is essential. A "review of the current situation" is closing the loop. Therefore also the "review of data" by suitable parties and the users groups with the related improvement input is a core aspect for the new planning of the next update cycle.

The key of a professional review process is a systematic approach and continuous process embedded into well defined update routines covering

- Basic or core technologies , e.g. power plants, refineries and water treatment units
- Dependent datasets derived from these core models
- Quality assurance processes

This process increases the overall transparency of data generation (with the additional benefit of independent verification), strengthens quality and credibility and leads to high confidence by the users of the data.

Feedback of stakeholders, demand determination of users and feedback from and into standardization processes and best practice guidelines is also important. Synergy of science and industry in database development is possible, if the different stakeholders stick to a modular system and their related responsibilities are understood and taken (Baitz et al. 2007).

3. Summary, conclusion and resulting actions

Specific feed and food data needs feed and food specific knowhow. However, any sector database must be usable and understandable in all branches. It is a long way from field to retailer, with environmental impacts from many different branches along the chain (see Figure 6). Therefore, isolated or sector specific data that are only understandable, communicable or exchangeable within the same industry branch are of less value for the LCA environment.

Figure 6. Data solutions in the network of branches.

Three key aspects, as an example of implementation for food and feed LCA databases, are provided in this publication. Based on scientific and methodological approaches, the GaBi LCA agrarian plant model covers all relevant LCA agricultural aspects. Parameterization provides the necessary flexibility and applicability to cover all agricultural plant products with a global applicability. The scope of the model is constantly developed and improved. For a scientifically sound, feasible and applicable scaling of dataset generation for use in industry, model parameters are prioritized and grouped and are determined via sequences of data collection. Using reasonable operating efforts the model provides consistent and very accurate results for various agricultural and plantation products and differentiated adjustable farming practices. A management and maintenance concept allows the use of the scientifically sound agrarian plant LCA models in practice and with broadly usable data. This management and maintenance concept includes a professional systematic review process with well-defined update procedures.

Ensuring the feasibility and practicability of a scientifically sound approach is essential for a professional database solution. Documentation of modeling approaches supports maximum transparency of various important aspects such as sources, quality assurance and review process. This approach is the basis and stepping stone towards regionalized (LCI and LCIA) dataset creation in the future. Similar approaches are to be applied for processing, by-products and other agricultural products with a special focus on the interconnection with other industry sectors.

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Proposal of a unified biodiversity impact assessment method

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ABSTRACT

The LCA community is moving forward in developing methods to address biodiversity in impact assessments and the required inventory data. However, biodiversity is a notoriously fuzzy topic. Underlying preferences and perspectives on biodiversity vary considerably among stakeholders. The method presented herein allows for both broad-brush and detailed assessments of critical processes. The goal is to enable LCA users to generate product-related biodiversity impact information and manage biodiversity along value chains. The method represents the impact category "biodiversity" in the UNEP-SETAC land use impact assessment framework. It is inspired by the method proposed by Michelsen (2008) in that it employs region-specific characterization models yet allows aggregation across regions. Preliminary results for a western European ecoregion show that landscape structure and pesticide use contribute to the biodiversity impact. Fertilizer input is relevant, but a low dose is tolerable.

Keywords: biodiversity, land use, impact assessment, method integration, landscape ecology, normative aspects

1. Introduction

Many international declarations and studies have built a strong case for protecting global and local biodiversity (e.g. UNEP 1992, MA 2005, TEEB 2010). The LCA community is moving forward in developing methods to address biodiversity in impact assessments and the required inventory data. In this paper we present an impact assessment method for terrestrial biodiversity. The method bears structural resemblance to the Michelsen (2008, 2013) method and takes advantage of expert judgment. It also aligns with the UNEP-SETAC land use LCIA framework (Milà i Canals et al. 2007, Koellner et al. 2013). Other LCIA methods for biodiversity, namely those that rely on statistical analyses of species inventories, lack the flexibility that our method offers. The statistical methods better suit unknown supply chains, i.e. background systems. Future unification of the various approaches may be possible.

2. Methods

Creating a biodiversity impact assessment category builds on previous work in developing an LCIA category for land use and land use change. Figure 1 shows a simplified schematic of land use change as described by the UNEP-SETAC land use framework (Milà i Canals et al. 2007, Koellner et al. 2013). A given land use impact can be described by Equation 1.

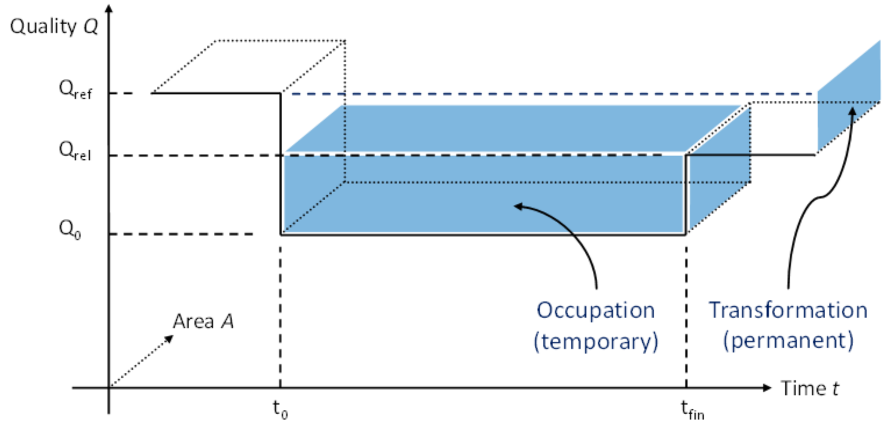

Figure 1. Simplified schematic of land use change as described by the UNEP-SETAC Framework. (Simplified from Milà i Canals et al. 2007, Koellner et al. 2013.)

$$\text{Impact} = \Delta Q \times A \times \Delta t \qquad \text{Eq. 1}$$

where:

Q = land quality (or other impact category)
A = area affected
t = time

In this description, the land use impact is the product of the land quality *Q*, the area of land directly affected *A*, and the duration time *t* of the impact. "Quality" on the y-axis represents applies to any impact category considered: in the UNEP-SETAC Framework, the y-axis refers to land quality, whereas here it refers to biodiversity quality.

The overall quantification of biodiversity is composed of two factors: an ecoregion factor and the actual assessment of the state of biodiversity at the given location. Biodiversity can be expressed via Equation 2 while the impact on biodiversity can be expressed via Equation 3:

$$\text{Biodiversity} = EF \times BP \qquad \text{Eq. 2}$$

where

EF = ecoregion factor
BP = biodiversity potential

$$\text{Biodiversity impact} = EF \times (1 - BP) \qquad \text{Eq. 3}$$

The biodiversity potential corresponds to a point in a multidimensional parameter space. The user obtains the function's value by entering a handful of parameter values x_i into (an ecoregion-specific version of) Equation 4. The scalar result can be interpreted in a similar manner as a potential function in field theory (e.g. a pressure potential that forces a fluid through a porous medium). The parameters are specific to each ecoregion, as is their combination in the biodiversity potential function. The values of these parameters are specific to processes in each ecoregion.

2.1. Ecoregion factor

The ecoregion factor is based on globally acknowledged biodiversity parameters, including total species richness, endemism, and vulnerability. Pertinent data, or proxies for the data, are obtained via online databases such as the World Wildlife Fund's WildFinder (WWF 2006). Michelsen (2008) does not call it an ecoregion factor but describes essentially the same methodological element as the combination of ecosystem scarcity (i.e. inverse area, or “smallness”) and ecosystem vulnerability (i.e. inverse pristine fraction remaining, or “anthropogenic encroachment”). This factor makes impacts comparable across ecoregions by assigning a weighting factor to each region. We use a similar but refined system, in which the ecoregion factor is calculated using the area of a given ecoregion, combined with the species richness, endemic species richness, and conservation status. (Brethauer 2013) The calculation details of the ecoregion factor are beyond the scope of this paper; this paper focuses instead on the biodiversity potential functions. (For calculation details, see Brethauer 2013).

Each data point is processed through a function to yield a contribution. The contributions are combined through compromise programming, which uses a behavioral model to aggregate the various ecoregions and compute the ecoregion factor. Table 1 lists sample input data and corresponding ecoregion factors. Ecoregion factors typically range from 0.4 to 2.2. Approximately 75% of all ecoregions are assigned a factor between 1.2 and 2.1, indicating that most biodiversity impact potentials would be doubled (at most) after applying the ecoregion factor.

Table 1. Sample ecoregion factors and their required data inputs.

Code	**Ecoregion name**	**Area [km²]**	**Total species**	**Endemic species**	**Conservation status[a]**	**Ecoregion factor**
AT0117	Madagascar Lowland Forest	111,760	509	230	1	1.99
NA0612	Northern Canadian Shield Taiga	617,319	182	0	3	0.58
PA0445	Western European Broadleaf Forests	493,836	381	0	1	1.37

[a] Conservation status is a measure of threat to an ecoregion assigned by the WWF.

2.2. Regional biodiversity parameters and contribution functions

Regional biodiversity parameters form the building blocks of the biodiversity potential functions. A contribution function for each parameter links the value of the parameter to a biodiversity contribution value. The contribution values lie between zero and one, where zero corresponds to the minimum contribution and one to the maximum contribution. The closer the contribution value is to one, the closer the considered state of biodiversity is to “good”. The term “good” is left deliberately undefined to accommodate varying conservation goals across ecoregions.

Picking appropriate parameters and defining the respective contribution functions require the developer to review the literature and to conduct a few expert interviews. National biodiversity strategies serve as a good starting point since every country participating in the Convention on Biological Diversity is required to have such a strategy document (UNEP 1992); reports from the International Union for Conservation of Nature (IUCN) and WWF reports are also good resources. The set of chosen parameters should include a mix of descriptive and management parameters, also known as pressure and state indicators, respectively, in the Driving-Force, Pressure- State- Impact- Response (DPSIR) Framework.

The expert interviews consist of a qualitative and a quantitative discussion of the parameters’ relation to biodiversity. First, the qualitative discussion: the developer and the experts being interviewed identify the general form of each contribution function. The parameter value is measured on the x-axis and the state of biodiversity on the y-axis. A large parameter value, indicating a good state of biodiversity, would be reprepresented by a point along the curve in the upper right corner. A factor detrimental to biodiversity would be indicated by a downward slope along the x-axis.

Second, the quantitative discussion: once the general shape of each function has been determined, a mathematical description of the curve can be developed. Each parameter x_i is related to its biodiversity contribution y_i through the function $y_i(x_i)$. A few general equations, each with some extra variables for curve-fitting, usually suffice. Most contribution functions thus far have been straight lines, s-shapes or u-shapes (upward or downward). The specific equations, graphs, and critical points are presented to the experts being interviewed and adjustments are made when necessary. The curve can also be critiqued by a wider audience, e.g. in a webinar or via an online questionnaire, to avoid overdependence on a few selected experts (this has not been done yet in our ongoing project).

2.3. Regional biodiversity potential functions

Once the relevant parameters for a given ecoregion have been identified and their respective contribution functions have been determined, the biodiversity potential function for the given ecoregion can be constructed. The biodiversity potential function combines *n* parameter contribution functions and normalizes the sum to the (0, 1) interval, as shown in Equation 4.

$$BP = \frac{1}{n} \times \left[y_1(x_1) + y_2(x_2) + y_3(x_3) ... + y_n(x_n) \right] \qquad \text{Eq. 4}$$

Equation 4 would be appropriate for a situation in which the parameters impact biodiversity independently from one other and in which all parameters have the same degree of influence. If this is not the case and instead some parameters are deemed more important than others, the more important parameters can be given a higher weight (greater than 1/n) and others a correspondingly lower weight.

When the interaction between parameters is more complex, two or more contributions functions can combined in other ways. They could be multiplied together so that both y-values must be large to reach a large total contribution. Instead of multiplying the functions, a fuzzy intersection can also be suitable in such a situation. When parameters compensate each other's contribution, a fuzzy union may be the best operational choice.

The result *BP* is the biodiversity potential function for a given ecoregion. It uses parameters as defined above as inputs and produces a scalar biodiversity value as the output.

3. Results

Parameters for the ecoregion PA0445 Western European Broadleaf Forests have been derived from the German National Strategy on Biodiversity (BMU 2007) and from expert discussions. Overall, nature conservation in Germany aims at producing and protecting low-intensity cultural landscapes (rather than pristine wilderness as in other parts of the world). The biodiversity contribution curves $y_i(x_i)$ are represented schematically in Figure 2.

Keeping the German context in mind, biomass removal is selected as the first parameter and is used as a proxy for management intensity. It is measured in multitudes of biomass production per harvest year. For example, a meadow from which all regrown biomass is harvested each year receives a parameter value of one. If half of the regrown biomass is removed, the parameter value is 0.5. A forest that is allowed to grow for 50 years and is then completely removed in a clear-cutting fashion receives a parameter value of 50 (probably not 50 because not all biomass remains to be harvested). These parameter values are the x-values of the contribution function. The function curve begins at large y-values, though not at the maximum. As the x-values increase, the biodiversity contribution rises to the top of the curve and then drops, eventually reaching a y-value close to zero (Figure 2, curve a).

The second parameter is fertilizer application, measured in kg N-surplus per hectare per year. Typically, ecosystems in the PA0445 ecoregion can cope with some N-surplus but an oversupply will allow fast-absorbing generalist plant species to outcompete the diversity of rarer niche species that are valued relatively high in conservation regulations. Fertilizer's contribution curve is s-shaped, starting at the high end of the y-axis before descending and leveling out close to zero at high x-values (Figure 2, curve b).

The third parameter is pesticide application. The experts interviewed agreed that every bit of pesticide is too much. The contribution function therefore is a curve that begins at the high end of the y-axis but declines sharply from the onset, eventually leveling out near zero (Figure 2, curve c). As a possible measurement that allows capturing of the variety of pesticides, eco-toxicity scores make for a good starting point (e.g. "comparative toxic units, ecotoxicity" or CTUe from the USEtox model).

The fourth and final parameter was chosen to be small landscape structures, measured in total covered area. In open landscapes, the structures could be walls made from loosely stacked stones separating fields, hedgerows, groups of trees or even single trees, strips of alluvial forest alongside bodies of water, etc. The contribution curve begins with low y-values, rises to the maximum, and drops again as the x-values increase (Figure 2, curve d). The curve levels out with small y-values. This curve illustrates that an overly structured landscape will not allow for as much biodiversity as a spacious landscape with some small structures sprinkled throughout. Conversely, too few small structures would characterize an overly homogenous landscape.

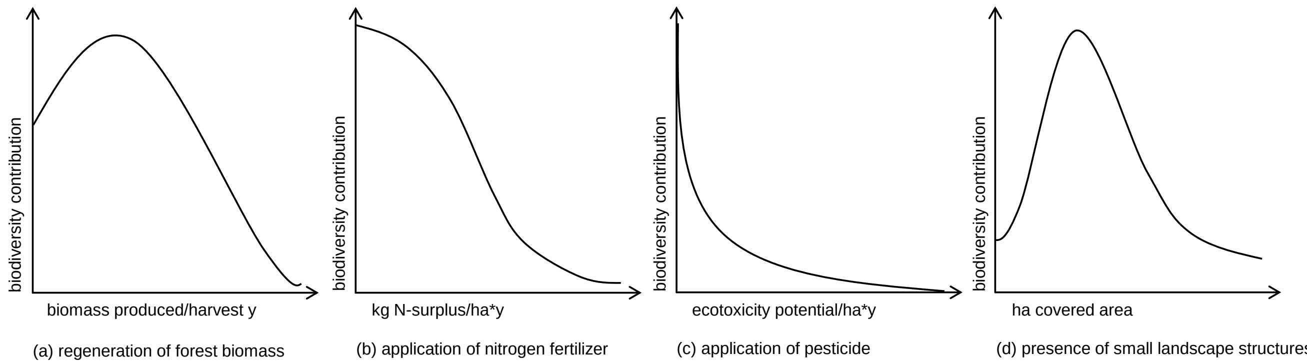

Figure 2. Biodiversity contribution curves of various parameters in ecoregion PA0445 Western European Broad-leaf Forests (schematic).

The functions were combined via summation. This very simple function combination process is to be understood as a first iteration and will likely be revised in future developments of the method.

4. Discussion

As far as we can tell at this point, the methodological framework generally functions as intended. It is possible to capture a range of differing interpretations of biodiversity as a conservation target. If desired, the potential field function can be replaced by a regionally agreed-upon method and integrated into the overall framework (hence the word "unified" in the title). With all input parameters on a continuous scale—as opposed to categorized by land use classes—differences in management practices that would otherwise go unnoticed can be captured and evaluated.

Our method trades consistency for comprehensiveness. Its structure tries to mimic an expert's impression of the overall state of biodiversity on an observed patch of land. Other methods that focus on particular measurable aspects of biodiversity (e.g. species diversity) are more consistent, however such methods (1) fail to capture biodiversity "as a whole" within a given region and (2) fail to acknowledge the varying interpretations of biodiversity across the globe. The method presented here is rooted in the conviction that a fuzzy subject like biodiversity requires an impact assessment capable of fuzzy analysis.

The reference state—a critical element for anything that uses the UNEP-SETAC land use framework—is implicitly defined as a byproduct of identifying the biodiversity parameters per ecoregion. For every parameter, there is a maximum biodiversity contribution. Anything below the maximum contribution is considered an impact. This means that the reference state represents a desired state of biodiversity as defined in national strategy documents. It acknowledges that whenever the term biodiversity makes the leap from a statistic to a political goal, it picks up a heavy normative load. We cannot define biodiversity as a conservation subject without naming what kind of biodiversity is actually desired. This defining task falls to governmental bodies, likely with the support of respected experts and conservation organizations.

Obviously, our biodiversity impact assessment method requires land use impact information that is typically only available for foreground systems. This challenge can be managed in two ways: (1) Archetypal combinations of parameter values can be developed that represent the majority of operations in a given ecoregion (e.g. "medium-sized cattle grazing operation in PA0445"), similar to archetypal emission profiles for industrial plants or vehicles. (2) Other more top-down biodiversity LCIA methods may be used for background system information, e.g. de Baan et al. (2012).

In regions with poorly defined conservation targets, hemeroby may serve as a surrogate since it is essentially a measure of naturalness. In absence of a specific "cultural" definition for biodiversity, such as the German one, naturalness seems like a good default. Michelsen and Coelho used hemeroby in a case study of the Michelsen method in New Zealand with moderate effort. As a composite index, hemeroby is not too different from the biodiversity potential described in this paper.

For more flexibility in the application of the method, distinguishing mandatory and optional parameters could help accommodate various types of activities. Parameters with general relevance for a given ecoregion would be mandatory while others would be selected from a list relevant only to certain activities.

5. Conclusion

This paper presents a method for biodiversity impact assessment in LCA. The first version of the method is currently being tested and is expected to undergo revisions over the course of the research project. Preliminary results, however, are promising and demonstrate the method's basic feasibility.

Inherent to the method is the concept of biodiversity as a fuzzy subject, both quantitatively and qualitatively. Depending on regional definitions and conservation goals, a regional biodiversity potential is defined. How much of this potential is conserved depends on regionally specific input parameters. The regional impact is weighted by the ecoregion factor to enable global aggregation. Other biodiversity LCIA methods are seen as contributors rather than competitors. This is especially true for regionally-specific methods, while the role of other methods as sources of background system information has been described as well.

To operationalize this method, the parameter list for PA0445 Western European Broadleaf Forests needs to be expanded and refined, and biodiversity potential functions for other regions need to be developed. Five case studies on various industry products are currently underway and are expected to offer further insight into the feasibility and accuracy of the method. Readers are encouraged to contact the corresponding author with criticism and other comments.

6. Acknowledgements

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Accounting for uncertainty in the quantification of the environmental impacts of Canadian pig farming systems

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ABSTRACT

The different environmental impacts of commercial pig production for two regions (Eastern and Western) in Canada were compared in a cradle to farm gate Life Cycle Assessment (LCA) using Monte Carlo simulations for statistical analysis. For the first time detailed production data from Canadian farms was used to quantify the impacts of commercial Canadian pig systems. The impacts were defined using 5 metrics – Global Warming Potential (GWP), Acidification Potential (AP), Eutrophication Potential (EP), Abiotic Resource Use (ARU) and Non-renewable Energy Use (NRE). Uncertainties in the model outputs were separated into two types: uncertainty in the data used to describe the system (alpha uncertainties) and uncertainty in impact calculations or background data which affects all systems equally (beta uncertainties). Using this method pig production in these two regions was systematically compared based on the differences in the systems (alpha uncertainties). The environmental impacts for impact categories GWP, EP, AP, NRE and ARU were found not to be significantly different ($P<0.05$) between Eastern and Western system, despite differences in the feed ingredients used and typical pig performance in the systems. The study quantified uncertainty systematically when comparing multiple environmental impacts measures in an LCA for pig systems for the first time.

Keywords: Uncertainty, Pig Production, Monte-Carlo Simulations, Canada

1. Introduction

Several recent LCA studies have quantified the environmental impact of pork supply chains in various farming systems using multiple environmental impact categories (e.g. Reckmann et al. 2013; Wiedemann et al. 2010; Pelletier et al. 2010; Nguyen et al. 2011). A smaller number, whilst quantifying a single impact category (namely Carbon Footprint) have introduced uncertainty in their results using Monte Carlo simulations (Thoma et al. 2011; Macleod et al. 2013). The aim of this study was to quantify the environmental impacts of typical pig production systems in Canada using multiple impact metrics, while integrating uncertainty in the LCA results using Monte Carlo simulations. An LCA was conducted to compare the environmental impacts of typical pork production systems in two regions of Canada; Eastern Canada (Ontario and Quebec) and Western Canada (Manitoba, Saskatchewan and Alberta). In 2012, production in these regions was ~13.5 million marketed pigs in Eastern Provinces and ~13.3 million finished pigs in Western provinces, a combined total representing ~98% of commercial pig production in Canada (Canadian Pork Council 2014). The two systems have several differences in pork production; including the feed ingredients used in typical diets, herd performance measures such as Feed Conversion Ratio (FCR) and mortality rates, as well as farm management practices such as finishing weights.

2. Methods

2.1. Model structure

A cradle to farm gate LCA was conducted to compare the environmental impact of pork production supply chains in Eastern (Ontario and Quebec) and Western (Manitoba, Saskatchewan and Alberta) Canada. The three main compartments of material flow in the Life Cycle Inventory (LCI) were the production of feed ingredients, the use of energy and materials in on farm pig production and the storage and land application of manure. The LCA modelled three separate stages in the pig production system; breeding (including suckling piglets), nursery (up to ~28 kg) and grower/finisher (from nursery to finishing weight). The functional unit of the LCA was 1 kg expected carcass weight at farm gate. For both regions typical diets for each stage of production were devised based on expert advice from Nutreco Canada: the Eastern diets were corn based with the western diets based on Wheat and Barley. LCI data for the major crop ingredients were taken from previous LCA studies of Canadian crops (Schmidt 2007; Pelletier et al. 2008). Using these diets and the herd performance data shown in Table 1 the nitrogen (N), phosphorus (P) and potassium (K) content of the resulting excretion was predicted. N retention in the finished pigs was calculated using an animal growth model using the principles of Wellock et al (2003); P retention was calculated using the method of (Symeou et al. 2014) and K using allometric relationships of body composition ((National Research Council 2012). All N, P, K not retained by the finished pigs was assumed to be excreted in faeces or urine and applied to land as fertilizer once losses during manure storage and spreading were accounted for.

The on-farm energy consumption data were based on studies of conventional pig housing systems in North America (Lammers et al. 2010). The mix of electricity generation in the LCA was the national mix for the Canadian grid (Statistics-Canada 2012); this was assumed for all Canadian unit processes in the LCA. The housing/manure model of important enteric emissions as well as those during manure storage and spreading was based on the principles of the IPCC guidelines for greenhouse gas inventories, using a tier 2 methodology. The model estimated the emissions of CH_4, NH_3, N_2O and NO during storage and application as well as the leaching of NO_3 and PO_4. The proportional mix of manure stored and spread using different techniques was based on the Statistics Canada records regarding the storage and application of swine manure (Statistics-Canada 2003; Beaulieu 2004). The calculated available N, P and K to the soil in the manure after land application were credited to the pig system by offsetting the need to apply synthetic fertilizers to land. As such the LCA modelled the net difference in emissions caused by applying manure instead of inorganic fertilizer, while accounting for the avoided burdens of fertilizer production. The proportional mixture of synthetic fertilizers applied to land in each region was derived from sales figures in a Canadian fertilizer shipments survey (Korol 2004).

2.2. Herd Performance data

The herd performance data used is shown in Table 1 and based on benchmark data from farms in Eastern and Western Canada. The Eastern data covered the performance of 73,880 sows, 1.47 million nursery pigs and > 1 million finished pigs, the Western data were based on 58,886 sows, 63,757 nursery pigs and 26,910 finished pigs. Using this data the model calculated the feed intake at each production stage per finished pig accounting for mortality at all stages and the flow of gilts to replace culled sows.

2.3. Environmental impact calculations

The environmental impact of the systems was quantified using the following metrics: Global Warming Potential (GWP), Eutrophication Potential (EP), Acidification Potential (AP), Non-Renewable Energy (NRE) and Abiotic Resource Use (ARU). GWP was quantified as CO_2 equivalent: with a 100 year timescale 1 kg CH_4 and N_2O are equivalent to 25 and 298 kg CO_2 respectively. EP, AP and ARU were calculated using the method of the institute of Environmental Sciences (CML) at Leiden University (http://www.leidenuniv.nl/interfac/cml/ssp/index.html). NRU was calculated based on the IMPACT 2002+ method. The impacts were calculated for a functional unit of 1 kg of pig carcass weight (CW) at farm gate, using a Canadian definition regarding yield (including head and trotters), meaning typical carcass yield was 80% (+/- 2%). The LCA calculations and Monte Carlo simulations were conducted using the SimaPro 7.2 software package.

2.4. Separating impacts

The results for each impact metric were assigned to the following material (and energy) flow categories to demonstrate their relative contribution to the overall impacts:

1) Feed: production of crops and additives, feed processing and transport. This category also includes the water consumed during housing.
2) Electricity: direct electricity consumption at the farm (breeding, nursery and grower/finisher stages) not including feed production, processing and transport.
3) Fuel: direct fuel consumption at the farms, not including feed production, processing and transport.
4) Housing: direct emissions of NH_3, CH_4, NO and N_2O from housing and manure storage
5) Manure: Emissions of NH_3, CH_4, NO and N_2O resulting from field spreading. This category also includes credits from replacing synthetic fertilizers.

Table 1. Mean, maximum and minimum values for key herd performance parameters

Indicator	East mean 2012	East min	East max	West mean 2012	West min	West max
Feed conversion Ratio Nursery	1.57	1.38	1.80	1.58	1.40	1.86
Feed conversion Ratio Grower/finisher	2.74	2.50	3.09	2.90	2.77	3.16
Average Daily Gain Nursery (g/d)	428	335	515	455	363	515
Average Daily Gain Grower/Finisher (g/d)	882	752	983	836	801	953
Start weight (kg)	6.32	5.50	7.30	6.22	5.40	7.00
Weight end Nursery (kg)	27.4	21.0	34.0	28.8	22.0	34.0
Weight Finish (kg)	123.6	118	130	118	114	126.6
Average weaned litter size (piglets)	11	9.7	11.6	10.8	10.1	12
Average weaning period (days)	20.0	18.0	23.0	20.6	18.6	26.5
Wean to eastrus (kg)	6.90	5.00	9.50	6.90	5.30	8.90
Litters /sow / year	2.45	2.20	2.55	2.43	2.30	2.50
Gestation feed/ weaned (kg)	28.8	25.0	34.5	28.0	25.6	34.5
Lactation feed / weaned (kg)	11.8	10	15.4	11.4	10	15.4
Creep feed / weaned (kg)	0.1	0.001	0.3	0.1	0.1	1.2
Still born	7.2%	4%	10%	7.6%	5.10%	11.9%
Post birth mortality	12.5%	6.20%	19.2%	13.6%	8.80%	20.4%
Nursery Mortality	2.8%	0.64%	7.50%	3.1%	0.90%	6.00%
Grower/finisher mortality	4.0%	1.50%	9.00%	3.0%	1.80%	5.90%
Sow Mortality	6.8%	3.60%	10.00	5.0%	2.00%	9.00%
Sow culling rate	36.1%	22.0%	58.0%	41.6%	27.7%	55.6%

2.5. Quantifying Uncertainty

A Monte Carlo approach was applied to quantify the uncertainties of the impacts in both systems (Leinonen et al. 2012). The LCA was run 1000 times, and during each run a value of each input variable was randomly selected from a distribution for this variable. Distributions were assigned to variables in the LCA based on the data available in each case. For example for major crops, yield data from 2010 to 2014 was used to estimate the average and typical ranges in the yield. In cases such as this or the herd performance benchmark data, triangular distributions were assigned where the position of the mean in the range did not suggest a normal distribution. Many of the distributions in generic unit processes taken from the ecoinvent database (e.g. transport emissions) came with log-normal distributions. The distributions for each impact metric were not assumed to be normal due to the effects of multiplying normal and non-normal distribution curves as part of the LCA uncertainty analysis. The outcome of the analysis was a minimum and maximum level of impact for each metric, which was used to evaluate the statistical significance of the differences between the Eastern and Western at 95% confidence. The uncertainties in the input variables were divided into two groups, namely system “alpha” and calculation “beta” uncertainty (Leinonen et al. 2012).

Alpha errors were considered to vary between systems, and therefore were taken into account in statistical analysis of the differences between the systems. For example, variation in the herd performance parameters seen in Table 1, used to calculate feed intake, were all considered to represent alpha errors. In contrast, beta errors were considered to be similar between the systems, and had no effect in the statistical comparison between the systems, e.g. the emission factor for N_2O from manure. The errors in the emission factors were associated with errors in the models used to generate them, and therefore considered as beta errors (Wiltshire et al. 2009; Leinonen et al. 2012). The error distributions of the emission factors in the manure model followed the IPCC (2006) guidelines as well the recent review by Liu et al. (2013).

3. Results

The LCA showed no significant difference ($P<0.05$) between Eastern and Western systems for ARU, AP, EP, GWP & NRE despite differences between both systems in terms of feed ingredients used and typical herd performance characteristics (Figures 1-5). As can be seen in figure 1-5 the range of results for most of the impact metrics do not reflect perfectly normal distributions.

For all impact categories the grower/finisher production stage accounted for >75% of impacts for both production systems. The nursery phase contributed not more than 11% of impacts for any impact category, with breeding no more than 16% for any impact category (Table 2). The production of feed ingredients accounted for the >90% of ARU and NRE and > 50% of GWP for both systems. Housing emissions from animals and manure storage were the largest contributors to AP in both systems (60 and 56 % respectively), housing and manure was also the largest cause of EP in the East (52%). However in Western systems feed ingredients were the largest cause of EP (49 %) due to the different feed ingredients used in the diet. The Coefficient of Variance (CV) for all impact categories was between 9 – 16 % for both systems. Manure application was associated with credits for ARU, GWP and NRE due to it replacing the production of synthetic fertilizers. For AP and EP the net effect of manure application was considered to increase emissions and leaching associated with these negative impacts.

Table 2**.** Summary of the environmental impacts of 1 kg pork carcass weight at farm gate for Eastern and Western Canadian pig systems (including source of impact in the systems by production stage)

	Abiotic Resource Use		**Acidification**		**Eutrophication**		**Global Warming Potential**		**Non Renewable energy**	
	kg Sb eq		kg SO_2 eq		kg PO_4^{3-} eq		kg CO_2 eq		MJ eq	
	East	West	East	West	East	West	East	West	East	West
Breeding	0.0012	0.0013	0.0088	0.0087	0.0022	0.0025	0.40	0.35	2.63	2.77
Nursery	0.0006	0.0008	0.0058	0.0079	0.0015	0.0022	0.27	0.28	1.69	1.92
Finisher	0.0062	0.0058	0.0492	0.0547	0.0129	0.0162	2.23	2.16	14.7	14.0
Total	**0.0081**	**0.0079**	**0.0638**	**0.0713**	**0.0165**	**0.0208**	**2.90**	**2.80**	**19.0**	**18.7**

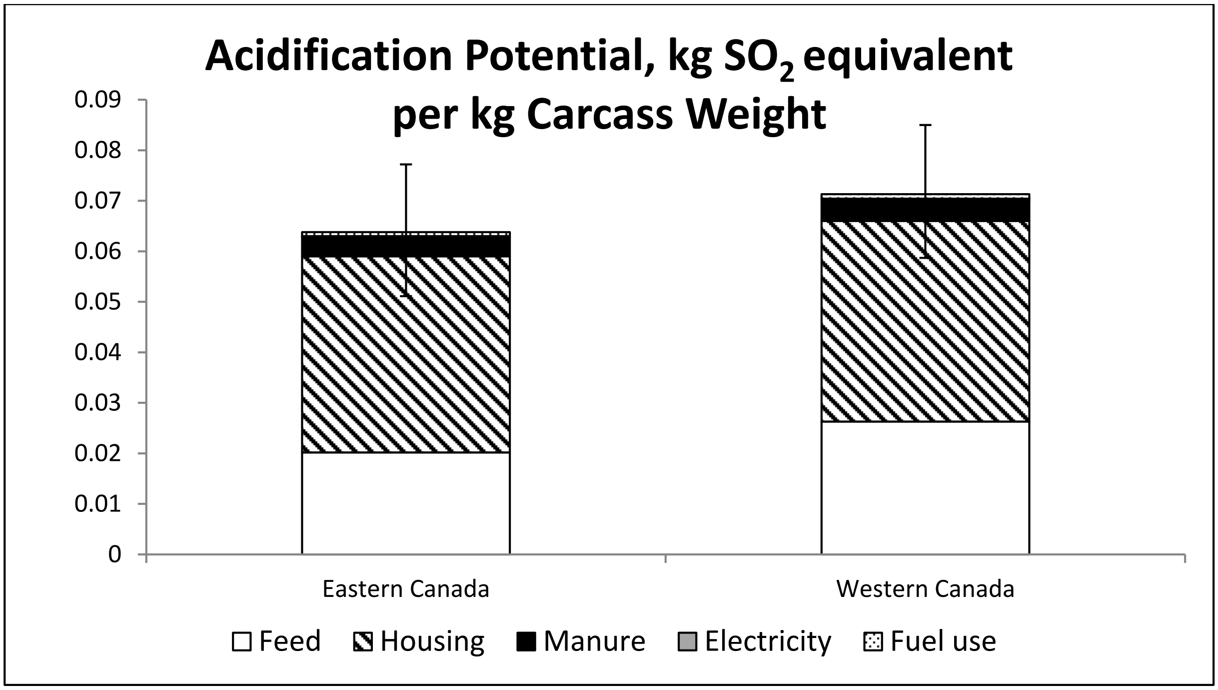

Figure 1. The Acidification Potential for typical Eastern and Western Canadian pig production. The error bars indicate standard deviations based on alpha uncertainties

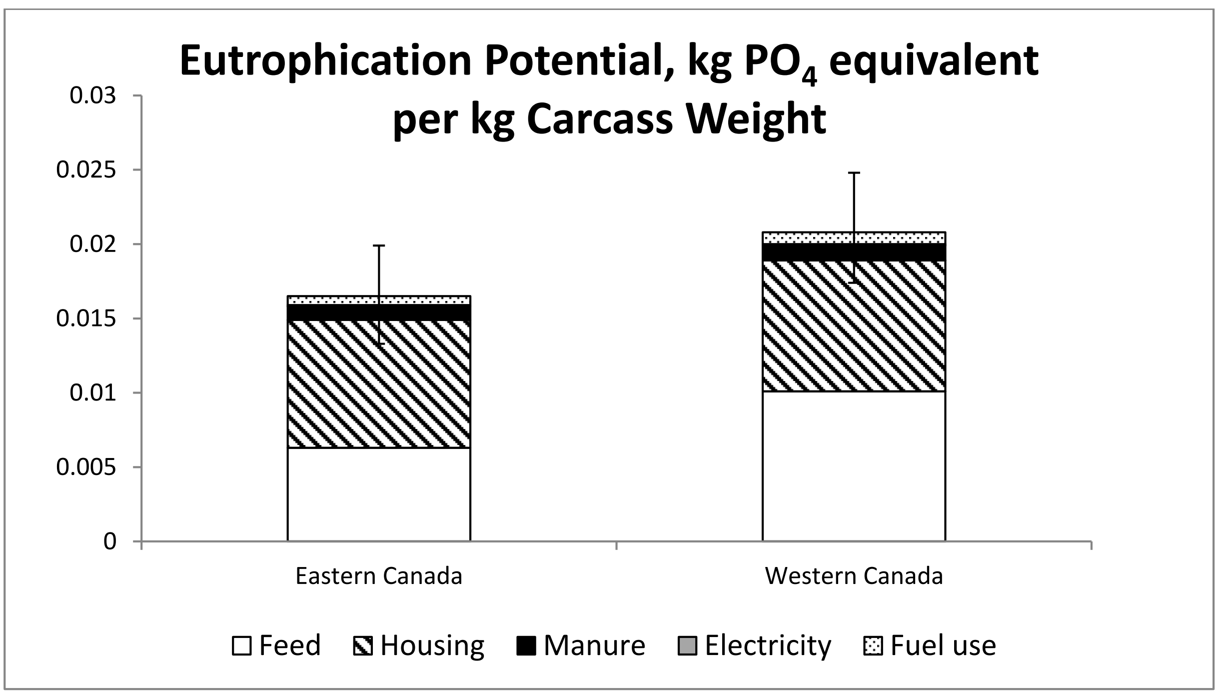

Figure 2. The Eutrophication Potential for typical Eastern and Western Canadian pig production. The error bars indicate standard deviations based on alpha uncertainties

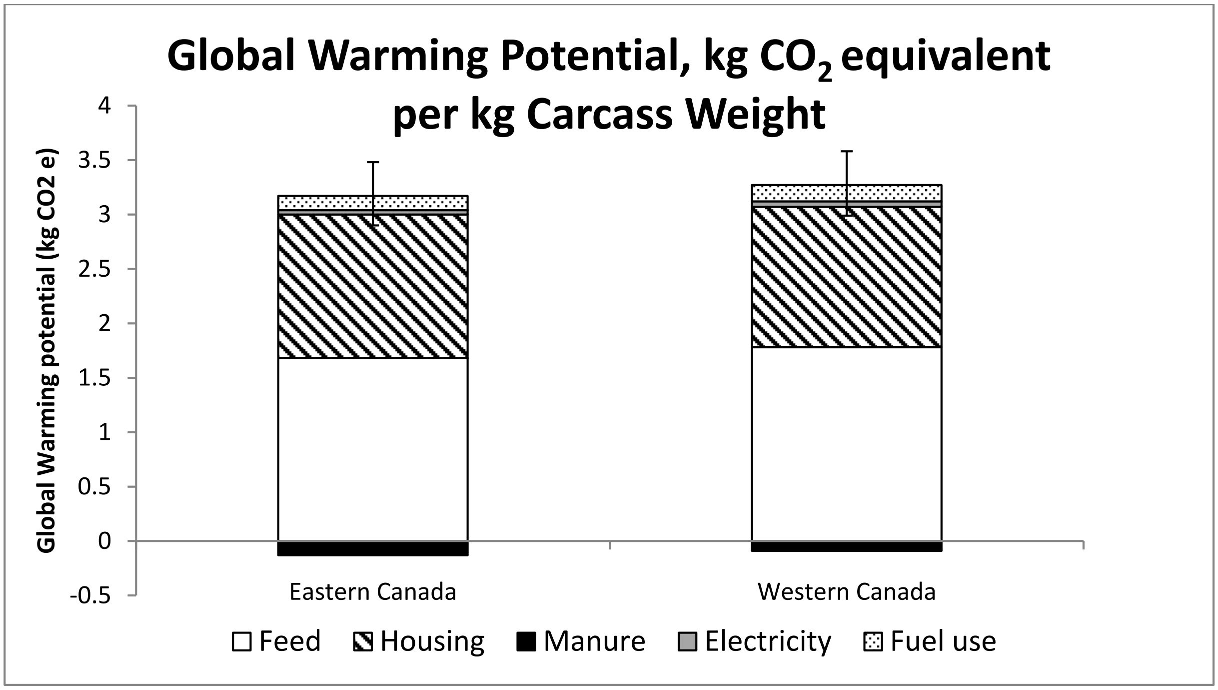

Figure 3. The Global Warming Potential for typical Eastern and Western Canadian pig production. The error bars indicate standard deviations based on alpha uncertainties

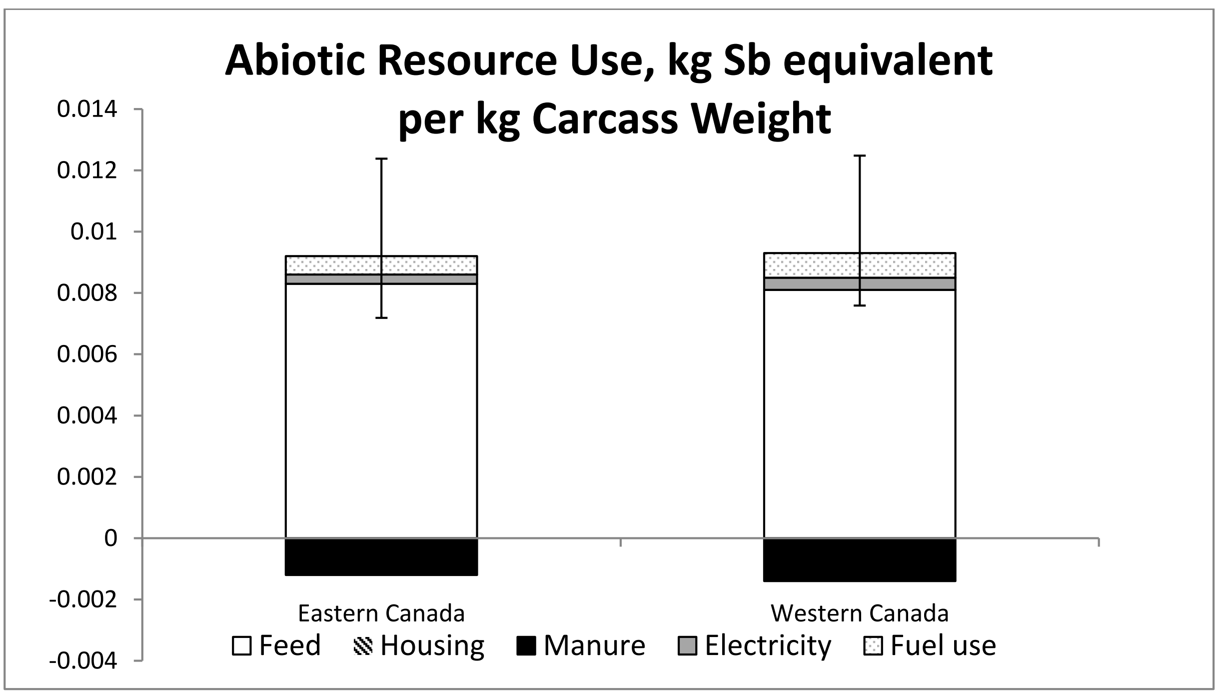

Figure 4. The Abiotic Resource Use for typical Eastern and Western Canadian pig production. The error bars indicate standard deviations based on alpha uncertainties

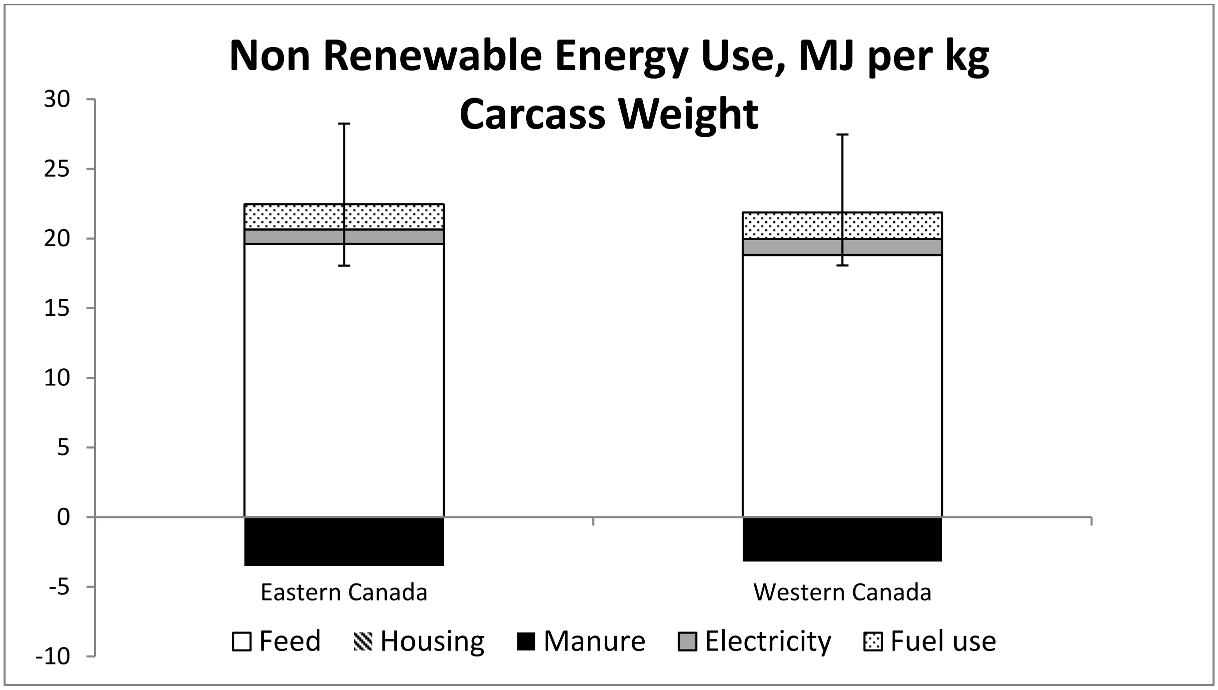

Figure 5. The Non Renewable Energy Use for typical Eastern and Western Canadian pig production. The error bars indicate standard deviations based on alpha uncertainties

4. Discussion

In all of the environmental impacts calculated, no significant differences between the Eastern and Western pig production systems were found. This was despite contrasting input data of the feed ingredients, herd performance and the manure applied to land as fertilizer. Although the Western system had numerically higher mean levels of GWP, EP and AP resulting from increased fertilizer requirements of major feed ingredients wheat and barley, in statistical comparison no significant difference in the environmental impact between the two systems was found.

The environmental impacts (particularly GWP) of pig production systems in many different countries have been quantified in previous studies, and summarized recently in the FAO report on Greenhouse Gases (GHG's) which result from monogastric livestock production (Macleod et al. 2013). This report sets out a new methodology to compare the results of pig LCA's in order to correct for differing assumptions and scope. Following this methodology, the results quoted for the three LCA's below are adjusted to represent the same FU and system boundaries as this study. The GWP results of this study do not appear different from the only previous peer reviewed carbon footprint study of Canadian pig production (Vergé et al. 2009), which reported 2.96 kg CO_2/ kg CW in Eastern systems and 2.85 CO_2/ kg CW. Previous LCA studies of US pork production have also reported figures within the range of uncertainty of this study for GWP of conventional pig production systems: 3.08 CO_2/ kg CW reported by Pelletier et al (2010) and 3.56 by Thoma et al (2010). It should be noted however that comparing results of LCA studies in a systematic manner is not feasible where the uncertainty range in those results is not reported. This is the case for most pig LCAs currently, especially for impact categories other than GWP.

Through separating different error categories in the input data, this study demonstrates a methodology for evaluating the differences between pig production systems in a consistent way. The use of Monte-Carlo simulations allows uncertainty analysis to incorporate the complexities of different types of distribution and incorporate covariance between parameters where this is suitable. Using this baseline model it will be possible to assess whether future changes to Canadian pig production systems such as an increased use of co products from biorefineries in pig diets or potential genetic changes can significantly reduce the environmental burdens of these systems.

5. Conclusion

The LCA examines the environmental impacts of commercial Canadian pig production, quantified for the first time using detailed industry data, multiple impact metrics and a systematic analysis of uncertainty. The study showed the importance of accounting for uncertainty when comparing the impacts of pig production systems. Such analysis has been lacking in previously published peer reviewed LCA's on pig systems. Monte Carlo simulations are a suitable tool for complex models such as LCA's to intrinsically account for uncertainty and allow for the complexity of non-normal distributions in the data. When modelling the systems stochastically the LCA found no significant differences ($P<0.05$) between pig production systems in Eastern and Western Canada for the environmental impact categories assessed. This was despite differences in the LCI of the systems compared regarding the typical feed ingredients used, herd performance data and predicted emission factors.

6. Acknowledgements

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The Sustainability Contributions of Urban Agriculture: Exploring a community garden and a community farm

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Abstract

This exploratory analysis of the potential of urban food production in cities of the global North is based on empirical examination of a community garden in New York City and a community farm in London, combined with secondary analysis of studies in six cities in the United States (US) and the United Kingdom (UK). Urban cultivation is promoted by environmental and food activists and organizations who may overstate the potential scale of urban food production: cities will be able to grow a maximum of about one-twelfth of a healthy plate for their inhabitants, and realistically much less. The main benefits of urban cultivation are social, and differ between urban and peri-urban farms. Capturing these benefits as contributions to the social component of sustainability represents a challenge for the development of social Life Cycle Assessment (LCA) or life cycle sustainability assessment.

Keywords: social sustainability, ecological education, food production, social LCA

1. Introduction

Community gardens and farms are a leading edge of the contemporary upsurge in urban cultivation. A sign of popularity is the endorsement of political leaders. For example, Michelle Obama planted a garden in 2009 with the help of school children--the first White House plot since Eleanor Roosevelt's World War II Victory Garden. The rise in interest is also indicated by a change in the status of urban cultivation, increasingly referred to as *urban agriculture* amongst academics (McClintock 2010). "Agriculture" projects a new frame and a larger scale than does "cultivation": agriculture is about food production, so that the horticultural gardens which dominate present urban cultivation are downplayed. The shift in perception raises research questions about the present reality and future potential of urban agriculture's output and sustainability. We address these questions based on case studies of a community garden and a community farm. The study is exploratory and descriptive, and addresses cultivation only as practiced in cities of the global North; the picture is quite different in the global South (Altieri 2012; Zezza and Tasciotti 2010).

There are pressing reasons for the interest in urban cultivation. Starting with the *macro* context, the world has a rising and increasingly urban population (UN 2011). There will be 2 billion more to feed by 2050, when 69 percent of our population of 9 billion will be urban, compared to 50 percent today. This growth is projected to increase food demand by 60 to120 % (Conforti 2011; Foley *et al*. 2011; Tilman *et al.* 2011; cited in Garnett & Godfray 2012). Progressive urbanization leads to loss of farmland (Seto *et al.* 2011). Between 1970 and 2000, the land equivalent of Denmark was converted from farmland to urban settlement. The projection for 2000 to 2030 is the equivalent of Mongolia, about 36 times the area of Denmark. Urban growth is also associated with tropical deforestation (DeFries *et al.* 2010). To exacerbate the problems, climate change is projected to result in farm yield loss (IPCC 2014; USDA 2013). Although there is debate around how large the loss may be, there is agreement that food security is one of the principal concerns humanity must address in the context of global climate change. The US is currently the world's 3rd largest food producer and its largest food exporter (FAO 2013) yet the US Department of Agriculture projects the yields of major US crops to decline by mid-century due to rising temperature and precipitation extremes. Thus agriculture faces a number of challenges in raising sustainable production levels. Many hope and argue that increasing production in cities and their suburban and exurban peripheries can contribute to meeting these challenges.

The institutional or *meso* context for rising interest in urban agriculture includes the environmental movement and related food consumption campaigns (*i.e.,* organic, locally-sourced, healthy and sustainable diets). Community gardening and farming evoke a cultural orientation different from that of traditional urban allotments. Allotments were in large measure institutionalized as compensation for the land clearances involved in the emergence of industrial agriculture in the late 18th--early 19th centuries in Northwestern Europe (see Fairlie 2009). In the UK, statutory allotment sites receive protection under the Allotment act of 1925, although there are fewer safeguards for private and temporary sites (RCEP 2007). Contemporary community gardening and farming represent a more recent movement, arising in the late 20th Century largely as neighborhood mobilizations to reclaim deteriorating open lands in post-industrial cities in North America and Europe.

It is at the local or *micro* level that urban agriculture has become a force for social change. In the context of this paper, "urban" refers to metropolitan areas (cities, suburbs and exurbs), while "cultivation" refers to controlled growing of any flora. The cases examined exemplify two common modalities of urban cultivation: one very small inner-city community garden and one larger but still small peri-urban community farm.

2. Methods

Data were collected through field observation, documentary and verbal information provided by informants, and accessing studies online. The main informants were the President of the West Side Community Garden (WSCG) and the Manager of the Sutton Community Farm (SCF). The WSCG in New York City was selected because one author had done volunteer work there since 2003 and began to study it in 2011; another author has visited it. The SCF in the southern outskirts of suburban London was selected because it had been the subject of a recent LCA (Kulak *et al*. 2013). All the authors took part in a site tour of the SCF in September 2013; one of the authors had worked on a similarly sized commercial market garden (known locally as a "smallholding") at that location some 50 years previously.

3. Results

3.1. Very small gardening: New York City's *West Side Community Garden*

The Garden began in the context of the massive 1970s Urban Renewal Program in the slums of gutted post-industrial cities (Martin 2011). The City of New York evicted occupants and razed tenement buildings in much of Manhattan's Upper West Side, leaving brown-field land available for redevelopment and gentrification (Wilson 1987). A high-rise condominium building was built on a site which included the future WSCG and another was awaiting capital investment. In the meantime the site became a dump for abandoned automobiles and other urban detritus. This dump site was transformed into a verdant garden in a spontaneous response by local residents to clean up a dangerous area in their midst that was also an eyesore. With construction imminent the neighborhood was assisted in saving this open space by the local Community Board and the Trust for Public Land. City government and developers acquiesced in part because community gardens enhance property values, thereby adding to tax revenues (while also, of course, adding to value for property owners). In an analysis of community gardens established in New York City between 1977 and 2000, Voicu and Been (2008:268) found that "gardens were located on sites that acted as local disamenities within their communities. . . after opening, gardens have a positive impact on surrounding property values, which grows steadily over time." The City administered a "sunshine test" and approved the site as a garden—with two stipulations for becoming untaxed land: that it would be open to the public and that it would pay for its upkeep.

The WSCG is located near the geographic center of New York City's Manhattan Borough. The land, 2/5 of an acre, is held by the Trust for Public Land and is governed by a board of officers elected annually from its 300+ paid members ("participants"). Membership is open to the public at a nominal annual fee.

Only about one-third of the Garden's space is used to grow food. Each gardener gets one raised bed of 30 ft^2. Gardeners reported that they do not grow much food—enough fruit and vegetables (f&v) for several meals a week over the late summer harvest period. "I just grow some nibbles," one said. Several informants related that growing food is not the main reason they gardened—rather, it was because they like to garden. Also, they reported that they like the cooperative aspects of the Garden and enjoy its ambience—a quiet, safe, public, and green island amidst Manhattan skyscrapers. Of the remaining 2/3 of Garden space 1/3 is devoted to horticulture. The final 1/3 is dominated by an amphitheater used for cultural productions.

The WSCG depends on a steady replenishment of labor to maintain compost bins and public areas, as well as to raise money. The Garden requires about $75,000 annually to operate. The bulk of the money goes to maintain pavements, towards insurance, and to purchase gardening supplies and tools. The labor required is skilled. This limits the available pool. Finding volunteer gardeners has been a general problem for community gardens. The largest pool of potential gardeners is women, mainly retired. New York City's gardens have declined in number since the mid-1980s largely due to lack of participation—many rely on one or two "tireless souls" (Tortorello

2012). The WSCG provides a range of cultural programs which attract thousands of visitors in the summer season who are potential sources of finance and labor.

3.2. Peri-urban small holding: London's *Sutton Community Farm*

This Farm comprises 7.1 acres, 3.5 of which are cultivated. It lies in the Borough of Sutton at the southern fringes of greater London, in what is termed "the green belt" in the UK planning system. It occupies green-field land but the soil is very poor. Until the 20^{th} Century lavender had been grown on the site as it can thrive in poor soils. The land use was changed as part of the mid-century drive to increase food production in the UK and took advantage of labor from a nearby camp for prisoners of war. Fifty years ago, the smallholding was operated by a family who lived there; it produced mainly high-value glasshouse crops, primarily salad vegetables and cut flowers sold via large wholesale markets in London, with high inputs including horse manure. There are now 500 m^2 of poly tunnels at the SCF providing for year-around production but it requires large inputs of compost, an expensive appetite for a non-residential farm with no manure-producing animals.

The SCF is London's largest community farm. It was started in 2010 with the blessing of Surrey County Council, which owns the land and collects ground rent. It operates as a co-operative and plans to offer shares within its local community. Similar social enterprises engaged in up-scaling local food production include *Urbivore* in Stoke (Williams 2013) and *Farmscape* in Los Angeles (Collins 2013).

The SCF is not solvent and there are no plans to make a profit. The goal is to make the Farm pay for itself and become a platform for activities for the local community; examples include making gardening experiences available to local school children and to disabled people. However, because of its location and the lack of public transport, a visit must be a planned activity. Salad crops are still the most profitable output, accounting for around 1/3 of income but only 1/7 of acreage, but the produce is consumed more locally than 50 years ago. About 3/4 of the Farm's produce is distributed to retail customers in "vegetable boxes"; this scheme currently has 142 customers, with a capacity for 350. The remaining 1/4 is sold wholesale, largely to local restaurants and cafes. The demographics of vegetable box customers reflect the local residential area: they are largely middle class. Many are seasonal customers who grow their own f&v and therefore buy much less in the summer. The Farm's unsold produce is collected by a local charity which makes soup from it. The two major expenses are compost (purchased from a local municipal site) for which haulage is the principal outlay, and water for irrigation.

The Farm's manager is a university graduate who used to be a chef and became interested in food security issues. He has organized an apprentice scheme at SCF. His view is that expertise in managing small-scale farms is generally lacking in the UK. The manager also organizes volunteer gardeners. The volunteers are diverse: some are employees of local businesses who are paid to work on the Farm as part of a Corporate Social Responsibility program.

Most of the Farm's tilled acreage is devoted to leaf crops. Surrey County Council requires the land to be turned over within two years, and so the Farm has no fruit trees. The Farm has not applied for Organic Certification, although the manager said that its production is "based on organic principles" and the Farm is open to anyone who wants to come and see for themselves. The Farm uses small tractors but most work is manual. The sole full-time employee is the manager. A "sustainable farming" apprentice grower is paid for three days per week. One grower is paid for one day per week. The vegetable box scheme has one employee working 3.5 days per week to deal with customers, and two drivers are employed, each for one day per week. Total paid labor is equivalent to 2.7 full-time workers.

3.3. Comparison between the two cases

The WSCG is an example of an inner city brown-field site coopted for social benefits; it depends on volunteer labor and contributions. The SCF is an example of a low-productivity green-field site which has transmuted into a social enterprise with some income and a paid work force. Both have outreach educational programs for their local communities. The Garden gets a large number of "walk-in" users and provides a sizeable cultural program, while the Farm produces considerably more food.

Using present formatting the very small Garden can grow the f&v portion of an annual *Eatwell* plate for about 3 people; the small Farm, for about 71 people. Extrapolation of these calculations, which are based on

FoodPrinting Oxford data (LCO 2012:28), show that it would require about 40 % of the total land areas of the New York and London urban agglomerations (UN 2011) to provide the f&v portion of an *Eatwell* plate for their populations. As f&v represent one-third of the plate, it would take more than **all** the land areas of the two agglomerations to grow full plates for their populations.

4. Discussion

4.1. Food production

Studies in 6 cities in the UK and the US have produced good and bad news for the potential of urban agriculture. The good news is that there is room for great expansion. The bad news is that this growth would come from a very low base and the maximum potential is low. Our initial analysis of research in Cleveland (Grewal and Grewal 2012), Detroit (Colasanti *et al.* 2010), London (Garnett 2001), New York (Ackerman 2012), Oakland (McClintock *et al.* 2013), and Oxford (*LCO* 2012) shows that an average of about 6 % of current f&v consumption is produced locally; this could be tripled to a maximum of about 18 %, which would still only represent about 6 % of an *Eatwell* plate.

This maximum potential is reachable only by overcoming some challenging conditions. Cities struggle today to maintain their current green spaces. In London the area of domestic gardens which comprise 25 percent of the land upon which f&v could be grown is declining. Between 1998 and 2008 "the area of plant-covered land fell by 12 % and the area of hard surfacing increased by 26 %" (Vidal 2011:3)—largely due to paving for car parking (Smith 2010). This is but one example of an imposing array of structural limits to urban food production: land, sustainability, labor and capital. With regard to land, urban settlement covers only 1 percent of the earth's land mass (FAO 2011) and it is expensive property subject to intense economic and political competition.

Another challenge to expanding urban agricultural production is the condition of brown-field urban soils. A study of lead contamination in vacant sites in Oakland found a high level of variability that must be considered when undertaking cultivation (McClintock 2013). Finally, there are sustainability questions about the advisability of converting urban green spaces (parks, greenbelts, etc.) to food production. These spaces already provide for carbon sequestration, urban cooling, biological diversity and social sustainability.

One touted technical solution for the lack of urban land is vertical farming or *z*-farming, for zero acreage (Despommier 2009). There are two major sustainability obstacles for high-rise *z*-farming: the energy required for artificial lighting to grow plants away from windows, and the industrial fertilizers needed to optimize yields from hydroponic production (Specht *et al.* 2013:10). These inputs obviously add substantially to the life cycle impacts of food production; see below. A technical issue for roof-top greenhouses is their integration with buildings, especially with recycling systems.

While structural obstacles will likely prevent the development of urban cultivation into urban agriculture, it has niche roles in local food production. One study has shown that it can make a significant contribution to the tables of low-income immigrants from agricultural backgrounds (Mares and Pena 2010). Another study has shown that it can make a substantial contribution to improving the diets of low-income persons with high rates of obesity and diabetes, and limited sources of fresh produce (McMillan 2008). However, in both cases the (volunteer) gardeners had free access to large plots of arable land.

4.2. Ecological sustainability

Underlining the caveats entered above concerning the potential for urban food growing, Kulak *et al.* (2013) have carried out an LCA study of the SCF which indicates that ecological benefits over the life cycle are localized rather than distributed along the supply chain. For example, reductions in greenhouse gas emissions are gained from appropriate choice of local crops that can substitute for foods grown in energy-intensive glasshouses or flown to the UK from overseas. A locally sensitive design approach is crucial in maximizing the potential for urban farming as a contributor to ecological sustainability.

Like all urban green spaces, community cultivation plots make contributions to local ecological sustainability—by providing natural habitats, improving soil quality, reducing soil erosion, and mitigating the city heat island effect (Bousse 2009; Brown and Jameton 2000; Cattell *et al.* 2008; Comstock *et al.* 2010; Ferris *et al.* 2001; Louv 2008; Pugh 2013; Relf 1992; SDC 2008; RCEP 2007; RHS 2011). Urban cultivation may make a greater contribution to carbon sequestration than other urban green spaces such as parks; Kulak *et al.* (2013) found this to be the case in their LCA study of the SCF. They may also reduce the run-off loss of rainwater exacerbated by the "concreting over" of cities and their environs (RCEP 2007); this is significant where, as in London, a principal aquifer may lie below the city itself.

Another contribution to ecological sustainability in which cultivation may outperform other urban green spaces is that it shelters biological diversity through a wide variety of flora—agricultural and horticultural. However, the contributions are dependent on local conditions; as a specific example, agriculture both supports and depends on the presence of bees to pollinate plants. The Farm at Sutton has three hives tended by a volunteer keeper, and is close to woodland and other commercial bee hives. Also, calendula, porridge, and ornamental flowers are grown to encourage pollination. In spite of being in the center of a large city, the WSCG has a good supply of bees from hives on nearby roofs and wild colonies in Central Park (Satow 2013). Thus, both the Farm and the Garden support bee populations by providing a diversity of flora—paralleling the practice of spacing ribbons of flowers amidst fields of crops in rural agriculture.

4.3. Social sustainability

The example of low-income immigrants illustrates that urban cultivation can make meaningful contributions to two major components of social sustainability—environmental justice and public health. However, it is in *education* that the WSCG and SCF make impressive efforts. The inter-generational principle of sustainability relies on ecological education. The Garden and the Farm are places where the practice of gardening is maintained and passed on to all age groups. The Garden reserves six plots for school children who participate in an ecology learning module during which they grow food. As follow-up to their experiences, children and their teachers have constructed several raised-bed plots in their schoolyard. In New York City, the number of registered school-based gardens has multiplied six-fold (Foderaro 2012). It is significant that most of the adult gardeners in the Garden have had previous gardening experience, many in their childhoods. The Sutton Farm operates a funded schools program, the *Green Grub Club*, in which pupils and after-school staff grow, cook, and eat vegetables. In addition, 16 students and their caretakers from a local school participate in a sponsored Disabled Farming Assistance program.

Informal education is a part of community cultivation as well. For example, whether WSCG should be primarily functional or "ornamental" is subject to on-going debate. There are four parties. "Foodists" make an environmental justice argument for converting flower space to f&v space in order to shorten the queue for beds (now a year) and provide more opportunity for those who are poor to grow their own. "Ornamentalists" make an aesthetic point for the beauty of flowers and its social psychological rewards, which are expanded to include a public health benefit. "Pragmatists" make an economic argument that flowers attract people who then contribute money and work to the Garden. "Ecologists" make a sustainability case for the biological diversity provided by flowers—and the bees that depend on them.

A latent result of the Garden's "great ornamental debate" is its contribution to the ecological knowledge of its participants. The debate takes place in informal interactions. Arguments are presented; this results in an exchange of information. Gardeners hear from each other about some of the complexities of food production and its relationship to sustainability. Several reported that they had learned about the four positions and "leaned" towards one or two of them. This communal learning is an example of the unique synergies that exist between ecological and social sustainability (see Martin 2013) and supports the argument here that small inner-urban plots mainly have social value. They add to conventional urban horticulture by being communal. The communality is a basis for development of social capital based on use value.

The social benefits of urban cultivation are felt mainly at a local level rather than distributed along a supply chain; they are outside the familiar framework of LCA. It is therefore questionable whether or not the benefits or relative disadvantages of urban cultivation can be captured by an approach based in LCA as currently conceived. The guidelines on social LCA (Benoit and Mazijn 2009) are "still very much in the developmental phase" (Paragahawena *et al.* 2009) and are in any case directed at detecting social "bads" in international supply chains. By contrast, urban cultivation can deliver social benefits which are then the driver for the activity. Similarly, the supply systems depend on inputs like imported compost, but also on affluent consumers—of vegetable boxes delivered to the doorstep in the case of SCF, and of cultural programs at WSCG.

Production, distribution and consumption are to be seen not just as a one-way flow of resources from supplier to consumer, leaving impacts in their wake, but as a channel by which benefits can flow back from the "consumer" (of food or land use) to the other agents in the chain (Clift *et al.* 2013). Adapting social LCA or life cycle sustainability analysis to this kind of case represents a new methodological challenge.

4.4. The role and future of urban cultivation

The difference between the two sites which are the focus of this study illustrates the big dilemma confronting urban cultivation. The WSCG is too small to provide a significant food output but has a high amenity value due to its location and accessibility. The SCF is large enough to provide a significant food output and some jobs, but its location makes its social value educational rather than amenity-based. Given the demand and price for land in "successful" cities such as New York and London, this distinction between urban and peri-urban land is likely to be persistent. The distinction is mitigated in distressed cities like Detroit where "abandoned houses, vacant lots and empty factories now make up about a third" of the city (Harris 2010:47).

Recognizing the difference between urban and peri-urban cultivation suggests a different approach to land provision, with planning and regulation on a regional or ecosystem basis. Being in the green-belt provides a measure of protection against development for the SCF. This kind of provision could protect existing food-growing land, particularly at urban peripheries, and could integrate food-growing spaces into new build and re-build. One vision for the UK is an increase from the current 4 % to 25 % of f&v urban and peri-urban production (*Growing Communities* 2012), representing 8 % of the *Eatwell* plate.

5. Conclusion

While structural limits prevent urban cultivation from becoming urban agriculture (at least in cities of the global North), there is a case to be made for it on the grounds of its contributions to ecological and social sustainability. Urban agriculture can produce little more than "nibbles" of food but it can contribute "oodles" to social-ecological sustainability. One can imagine an urban regional cultivation scheme in which there are more community gardens in the center, more allotments and domestic gardens in the suburbs, and more community farms in the exurbs. The WSCG and SCF cases illustrate that "To be a viable alternative in cities and compete with other land uses, the justification for urban agriculture must include the ecological and cultural function these systems offer, in addition to the direct benefits of food produced" (Lovell 2010:2516). Moreover, the co-benefits could include 'spill-over' of interest in local food to changes in other practices at household and community level, such as greater awareness of waste reduction, long-life product use and reduction in car dependency. How engagement in urban farming can act as a catalyst for wider practices of 'sustainable lifestyles' is a rich field for research.

Concerning the scope for the scalability of urban cultivation, perhaps we should be looking at food systems differently? What if there is a parallel with utilities in energy, and with the scope for distributed community energy systems? In this case food should be seen as a good currently supplied by oligopolistic intermediaries (retailers) from ever more consolidated primary producers (farmers), a state of affairs that needs reform for greater resilience - there are many parallels with energy generation and distribution. What if we had community farms that are hubs of services and production, with 'sub-stations' in residential areas (allotment tillers and keen back garden food growers) acting as the equivalent of a localized energy grid and generation system? As with decentralized energy, the aim would not be to achieve total self-sufficiency and grid-independence, but instead to boost system-wide resilience via redundancy, diversity and storage. After all, food systems represent flows.

The looming food security crisis will not be resolved by transforming urban cultivation into urban agriculture. Instead this will depend on meeting a daunting list of challenges: Reducing food waste, which accounts for up to one-third of production throughout the food chain (Kummo *et al.* 2012); shifting crops away from animal feeds and biofuels to human foods, which can increase global calories by up to 70 per cent (Cassidy *et al.* 2013); adopting *sustainable intensification*, in which productivity is raised without increasing environmental impact and without using more land (Garnett and Godfray 2012); and shifting to ***sustainable*** *Eatwell* plates on the consumption side (Harland *et al.* 2012; Macdiarmid *et al.* 2011; Thompson *et al.* 2013). Urban cultivation can have a small but significant role in such an evolving sustainable food system, as a provider of some local food, and as a catalyst for socio-economic sustainability.

Science has provided us with the ecological metrics needed to specify the requirements of food sustainability. The big gap in our knowledge is an understanding of the *social* processes and practices necessary to provide for food security. *Society* is the neglected child in the sustainability family. Urban cultivation offers a unique venue for assessing social sustainability and its synergies with ecological sustainability. The needed calculations pose a challenge to social LCA methodologies.

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A comparative assessment of greenhouse gas emissions in California almond, pistachio, and walnut production

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ABSTRACT

A process-based life cycle assessment (LCA) model was constructed for almond, pistachio, and walnut production in California. Agrochemical inputs, mechanized operations, soil processes, geospatial variation, and biomass accumulation were explicitly modeled based on technical reports, economic cost-and-return studies, field data collection, and grower interviews. Mean annual greenhouse gas (GHG) footprints for a typical hectare of orchard, from nursery to hulling/shelling facility gate, were calculated at 4260 kg CO_2e ha^{-1} yr^{-1} for almond, 3480 kg CO_2e ha^{-1} yr^{-1} for pistachio, and 4050 CO_2e ha^{-1} yr^{-1} kg for walnut. These results can be expressed by orchard product (nut kernel) as 1.76 CO_2e kg^{-1} for almond, 0.95 CO_2e kg^{-1} for walnut, and 3.83 CO_2e kg^{-1} for pistachio. Variations in biomass accumulation, yield and orchard lifespan between these crops result in different total life cycle emissions and potential management options for net GHG reduction and credit opportunities under California GHG cap-and-trade legislation.

Keywords: perennial cropping systems, orchard, greenhouse gas footprint, almond, walnut, pistachio

1. Introduction

Though LCA has been applied to a wide variety of food production systems, orchards have been examined relatively infrequently, with studies focusing on apple (Milà i Canals et al. 2006; Mouron et al. 2006; Page et al. 2011; Hester & Cacho 2003; Alaphilippe et al. 2012), kiwi (Xiloyannis et al. 2011), and citrus (Coltro et al. 2009; Mordini et al. 2009; Beccali et al. 2009; Beccali et al. 2010). Walnut production has also been examined although primarily in the context of timber production (Cambria & Pierangeli 2011). Many of these studies examine a single year for a production system, and thus do not consider the entire orchard life cycle, which includes orchard establishment, tree growth, and eventual removal (Bessou et al. 2012). Also, these studies generally do not consistently address the flow of carbon and nitrogen through the orchard and the woody biomass generated from orchard systems.

This study characterizes the greenhouse gas impacts of typical almond, walnut, and pistachio orchard production systems in the U.S. state of California using a comprehensive process-based life cycle assessment model that specifically accounts for carbon and nitrogen flow in biomass accumulation and fertilizer application. These highly industrialized agro-ecosystems are of great economic and environmental importance, occupying more than 500,000 ha of California agricultural land and yielding more than 83% of world almond production, 46% of walnut production, and 37% of pistachio production (USDA Office of Global Analysis 2013). Commercial orchards in California's Central Valley demand significant agrochemical, water, and fuel inputs throughout their productive lifespans (Beede et al. 2008; Connell et al. 2012; Grant et al. 2007; Micke & Kester 1997). Irrigation accesses groundwater via on-site pumps and surface water via the California Aqueduct system and other surface water transport infrastructure, entailing significant energy inputs for onsite and upstream pumping. Due to its relatively high input intensity, the California nut industry is responsible for significant emissions of greenhouse gases (GHGs) and other atmospheric pollutants.

However, perennial cropping systems such as almond, walnut and pistachio orchards have the potential to sequester carbon in soils and biomass as a consequence of their long life cycles and high biomass production (Kroodsma & Field 2006). In California, much orchard waste biomass is used to produce electricity at regional electricity generation plants, widely distributed in California (Wallace & Leland 2007). The potential for sequestration versus emissions offset through use of waste biomass as an energy feedstock is dependent on management characteristics, orchard lifespan, and other factors. We examined the effects of various biomass fates on net orchard GHG and energy consumption footprints, accounting for potential GHG credits from biomass-based energy production and temporary storage in standing biomass under both a business-as-usual scenario based on estimates of current practices, and a maximized energy production scenario assuming that all biomass co-products are directed to energy generation.

2. Methods

2.1. Model Development

This orchard LCA model was developed originally for analysis of almond production, and subsequently adapted for analysis of walnut and pistachio production. The general model structure remained the same, with multiple years of the orchard life cycle treated separately in order to account for changing input requirements and yields as trees mature, and model subunits for calculation of irrigation energy requirement, soil nitrous oxide emissions, and fuel combustion emissions were identical. The major differences between the three systems are in agrochemical and water input requirements, fuel use in field operations, typical yield per hectare, planting density and biomass accumulation, orchard lifespan and length of time before full bearing, and geospatial distribution and spatial relationships with irrigation infrastructure.

Where data were not available, particularly in the case of pistachio and walnut hulling and shelling operations, emissions from pistachio and walnut orchard floor soils, and pistachio nursery sapling production, data were substituted from the almond model (Kendall et al. 2012). Although data on the major system drivers of lifespan, nutrient management, irrigation, and yield are complete for almond, walnut, and pistachio, data collection for some aspects of walnut and pistachio production remains ongoing and will be included in further model iterations and published as stand-alone LCAs of walnut and pistachio production. The results presented are calculated from generalized, statewide data; however, the model framework is also usable for case studies of individual orchard operations.

2.2. Data Collection and Sources

To date, the most complete data were available for almond production. Data on input requirements, field operations, planting density, orchard productive lifespan, and irrigation systems were obtained from University of California Davis (UCD) Cost and Return studies. These studies document annual crop production costs for various California crops, by inventorying typical inputs and cultural practices on a regional basis up to the farm gate (M. A. Freeman et al. 2003; M. W. Freeman et al. 2003; Duncan et al. 2006; Connell et al. 2006; Freeman et al. 2008; Duncan, Verdegaal, Holtz, Doll, K. A. Klonsky, et al. 2011; Duncan, Verdegaal, Holtz, Doll, K. M. Klonsky, et al. 2011; Connell et al. 2012; Beede et al. 2008; Grant et al. 2007). The most recent available data was used, ranging from 2003 – 2012 for almond, 2007 for walnut, and 2008 for pistachio.

These Cost and Return studies are developed based on data collected from growers, orchard managers, and UC Cooperative Extension farm advisors through surveys, interviews, and focus groups. They provide a picture of the typical nutrient, pesticide, fuel, water use, equipment use patterns (including equipment type and hours of operation), and annual yields for an orchard under a particular irrigation system type in a particular growing region (Sacramento Valley, San Joaquin Valley North, and San Joaquin Valley South). In this LCA, the most conservative available regional data from the studies is used. The term “conservative” here refers to the use of the highest typical input values, which reduces the risk of underestimating inputs and associated GHG emissions.

Because growing region and irrigation type can affect key factors such as yield, results are calculated as a weighted average based on the number of hectares of orchard per region, and the type of irrigation system used; flood, micro-sprinkler, sprinkler or drip. Regional distribution information on almond irrigation methods and the proportion of groundwater versus surface water used by growers was obtained from survey data commissioned by the Almond Board of California to develop a sustainability program for the state’s almond growers. For walnut and pistachio, these data were obtained through interviews with UC Cooperative extension crop specialists and farm advisors.

One shortcoming of using these studies to inventory inputs and operations is that custom operations, those operations conducted by contractors rather than the orchard owners and managers, are tracked only as a cost in these studies, omitting information such as the hours of equipment operation and chemical or fuel inputs associated with these operations. To fill these and other data gaps in the LCA model, additional data were obtained through surveys of businesses and individuals involved in nut production. Surveys were administered to nursery operators, growers and their orchard managers, custom harvest operators, and orchard clearing operators. These surveys were conducted by both on-line survey and in-person interviews. In-person interviews were conducted to collect data for specific aspects of an operation, particularly equipment use and time needed for various tasks. No primary survey data for individual respondents are reported in this article to protect the anonymity of cooperating individuals and businesses, but wherever possible aggregated or composite results from surveys are provided.

Data for hulling and shelling were collected for almond through surveys and interviews of facility operators, as reported in (Kendall et al. 2012). In absence of similar data for walnut and pistachio, almond values were used. Further data collection is ongoing, but given the relatively small contribution of post-harvest operations to the overall GHG footprint of nut production (Figure 5), significant changes to overall net GHG footprints are not expected from the use of walnut and pistachio specific hulling and shelling data.

2.3. Goal and Scope Definition

The goal of this project is to conduct process-based LCAs for typical commercial California almond, walnut, and pistachio production to estimate the GHG emissions associated with production activities. Operations and inputs that contribute the most to total emissions and energy (i.e., ‘hotspots’) over the orchard life cycle are also identified to assist growers and policy-makers in targeting the highest-emitting or highest energy-using processes for reduction. The processes and operations included within the LCA system boundary are illustrated in Figure 1 below.

The units of analysis are one hectare of orchard assessed over a time horizon equal to the productive lifespan of the orchard plus one year for orchard clearing and fallow: 26 years for almond, 36 years for walnut, and 61 years for pistachio; and one kilogram of nut-meat or kernel (obtained by dividing through by mean annual yield on a kg ha^{-1} basis) at the hulling and shelling facility gate. Both GHG emissions and yield vary as the orchard matures, so annual yields were averaged over the orchard lifespan including non-productive and fallow years in order to calculate a generalized value for GHG emission expressed on a per kilogram yield basis. This functional unit can be simply converted to nutritional units such as calories of food energy.

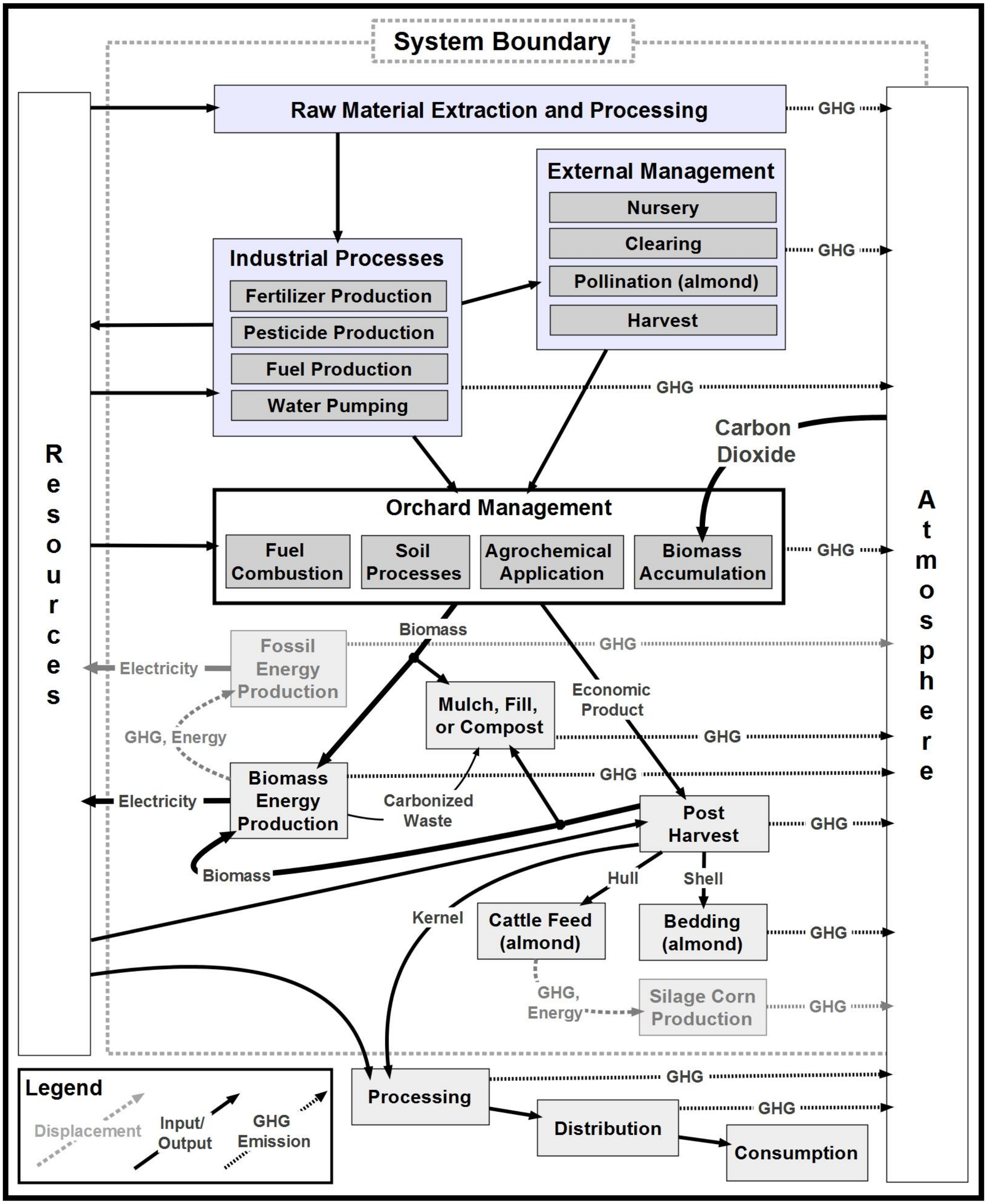

Figure 1. Orchard LCA system boundary. Transportation emissions, though accounted for at each stage, are excluded for clarity.

2.4. Model Assumptions

Model assumptions follow those reported previously for almond production (Kendall et al. 2012), and include the following:

- One percent of trees in each cropping system die and are replaced each year.
- The land occupied by the modeled orchard is assumed to be continuously planted as orchard, resulting in no net change in soil carbon over an orchard's life cycle. Though soil carbon changes over an orchard life cycle it is assumed that the soil carbon level at the time of orchard removal is identical to the soil carbon level following orchard establishment, due to loss of soil carbon during land clearing, tillage, and planting. Additionally, since potential carbon storage in soil is highly dependent on site-specific soil characteristics and orchard floor management (Six et al. 2004), accurate assessment of potential soil carbon sequestration in orchards on a statewide basis will require further field data collection and will be addressed in future publications.
- Different irrigation systems are modeled independently because they result in differences in both water input (including pumping and pressurization requirements) and direct and indirect nitrous oxide (N_2O) emissions from the field (estimated using IPCC Tier 2 methods). In terms of N_2O emission factors, the different crops are assumed identical and only differences between irrigation systems are accounted for.
- Agricultural equipment production is unlikely to have a major impact on the results of this analysis and is excluded - an exclusion common in other LCA studies (British Standards Institution (BSI) 2011).
- For purposes of comparison, each of the three orchard types is examined on a time horizon of 61 years: 1 pistachio life cycle, 1.71 walnut life cycles, and 2.40 almond life cycles.
- Inputs, especially nitrogen fertilizer and irrigation water (Figure 2), vary according to the nutritional needs of the different crops and the regional climate as well as over time as the orchard matures.

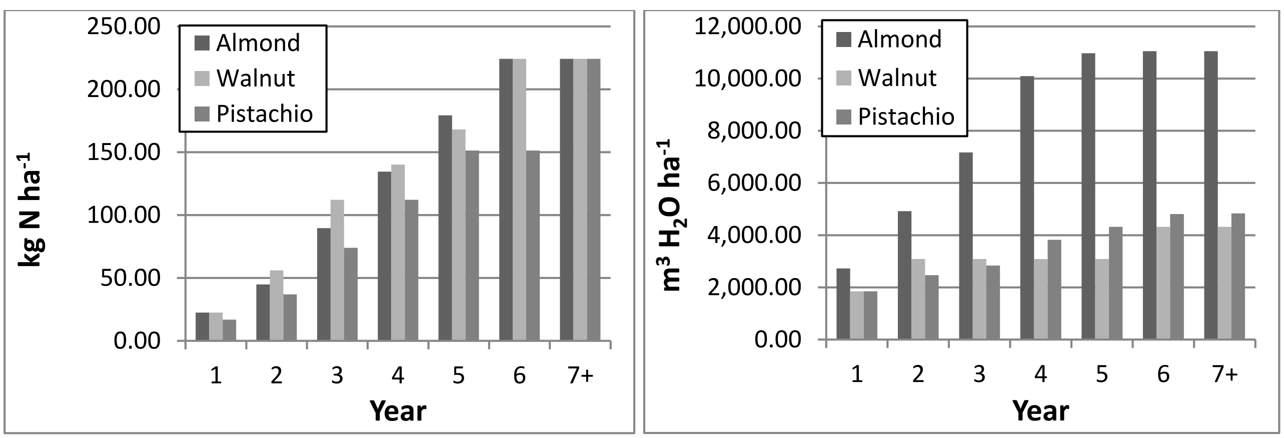

Figure 2. Variation in applied nitrogen and irrigation water by crop and year.

- Irrigation energy use is regionally dependent due to the spatial relationship with surface water delivery infrastructure (e.g., the number of California aqueduct pumping stations upstream of a given water diversion) and average groundwater depth, and thus the region of the Central Valley in which an orchard is located determines the energy requirement and the resulting GHG emissions embedded per unit applied water. A GIS-based model was used to generate a weighted mean irrigation energy value based on prevalent irrigation system types and spatial distribution of almond, walnut, and pistachio orchards. A detailed explanation of this model as applied to almond is available in (Kendall et al. 2012). Pistachio orchards are mostly concentrated in the southern Central Valley where groundwater is deeper and surface water delivery more energy intensive (Klein & Krebs 2005; GEI Consultants & California Institute for Energy and Environment 2010), while walnut orchards tend to be located in the northern Central Valley where surface water is gravity-fed and groundwater is closer to the surface. This results in higher typical irrigation energy use for pistachio and lower for walnut as compared to almond (Table 1).

Table 1. Almond, walnut, and pistachio irrigation characteristics.

Crop	Irrigation Energy	Main Irrigation Type	Mean Annual Applied H_2O
Almond	0.59 MJ m^{-3}	Microsprinkler	10307 MJ ha^{-1}
Walnut	0.40 MJ m^{-3}	Solid-set Sprinkler	4112 MJ ha^{-1}
Pistachio	0.83 MJ m^{-3}	Drip	4686 MJ ha^{-1}

2.5. Co-product treatment

As above, model treatment of biomass co-products including hulls, shells, prunings, and removed trees follows that reported in (Kendall et al. 2012). Some of the major differences between co-product treatment in almond, walnut, and pistachio are as follows:

- Biomass accumulation in almond was modeled according to a logistic curve, fit to measurements of biomass cleared per hectare obtained through collaboration with a Central Valley agricultural services firm. This method results in accurate calculation of total biomass per hectare at the end of the orchard productive lifespan (here 25, 35, and 60 years for almond, walnut, and pistachio respectively), but total biomass at any given year between establishment and clearing remains uncertain. Destructive sampling of individual trees of known age was used in , although actual biomass at end of life could only be estimated based on the logistic growth model (Agueron & Roberts 2013). In walnut, both destructive sampling of trees of known ages and measurements of orchard clearing biomass were used, giving accurate estimates of orchard biomass at all stages of growth.

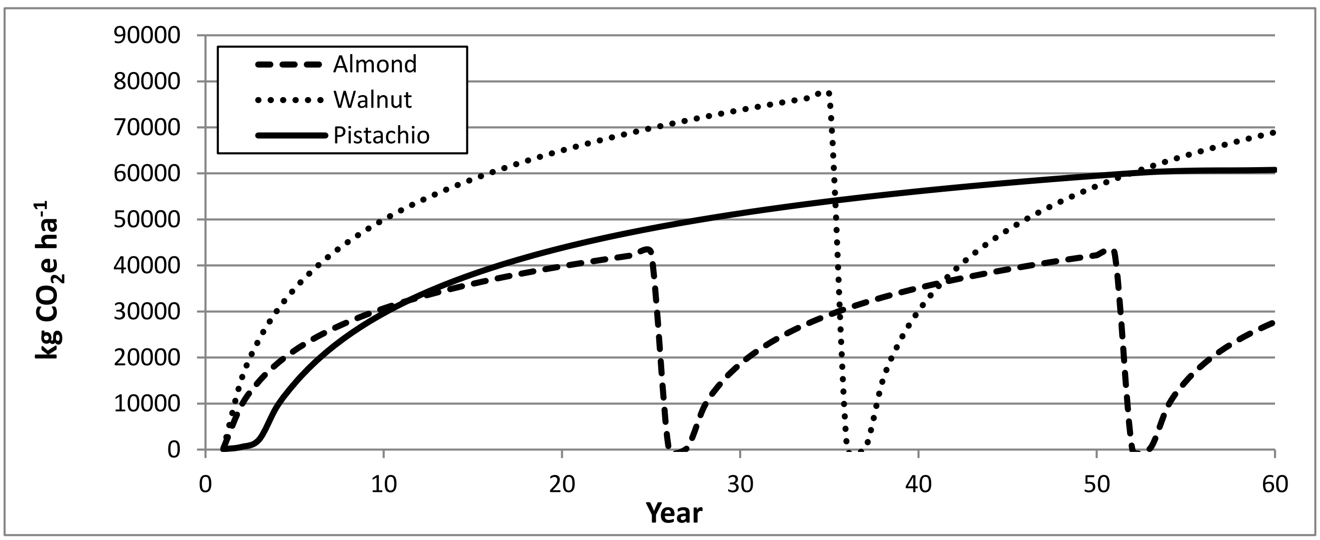

Figure 3. Biomass accumulation as CO_2e ha^{-1} in almond, walnut, and pistachio orchards over a 60 year time horizon.

- Pruning removal was calculated as a function of total tree biomass per hectare and pruning removal estimates (kg ha^{-1} yr^{-1}) from mature orchards. This base rate varied from 1060 kg ha^{-1} in almond to 1460 kg ha^{-1} in walnut and pistachio (Williams et al. 2008).
- Ratios of hull and shell to kernel (Table 2) were obtained directly from hulling/shelling facility records for almond, and from literature for walnut and pistachio (Monselise 1986).

Table 2. Mean annual yield and hull and shell to kernel ratios for almond, walnut, and pistachio.

Crop	Shell: kernel mass ratio	Hull: kernel mass ratio	Mean Annual Yield (kernel)	Mean Annual Yield (total)
Almond	0.45	1.56	1359 kg ha^{-1}	4091 kg ha^{-1}
Walnut	0.77	1.77	2631 kg ha^{-1}	9314 kg ha^{-1}
Pistachio	1.01	2.63	1375 kg ha^{-1}	6380 kg ha^{-1}

- Typically, almond hulls are used as dairy cattle feed while pistachio and walnut hulls are mulched. Specific information on shell fate was not available, so shells were assumed to be mulched for all three crops.

As reported in (Kendall et al. 2012), the rate of cleared biomass directed to energy production was reported as 95% for walnut and almond. In absence of specific data, the same rate was assumed for pistachio. The energy content of wood was obtained from (Wallace & Leland 2007), and a reasonably low estimate for power plant conversion efficiency of 0.25 was used to determine electricity generation offsets (Bain 1993). The equivalent emissions from a typical California grid electricity generation mix were considered to be displaced by the electricity produced from orchard biomass.

Each kilogram of biomass generates approximately 2.57 MJ of electricity after being dried in-field and at the power plant to approximately 30% moisture. A conservative value of approximately 25% energy conversion efficiency was assumed for California biomass power plants. When almond hulls are fed to cattle they are assumed to displace roughage, assumed to be silage corn, on a one-to-one mass basis. Displacement of electricity and silage corn production by use of orchard biomass co-products results in avoided GHG emissions, and is used to calculate GHG credits through system expansion to include energy generation and cattle feed production (Marland & Schlamadinger 1995). LCI data were obtained from EcoInvent and PE databases, accessed via GaBi software packages (Centre 2008; PE International 2009).

2.6. Temporary carbon storage

Temporary carbon in tree biomass is a function of marginal biomass accumulation on an annual basis as well as total storage time (orchard lifespan). Temporary carbon storage is a special case of emissions timing, where a removal of CO_2 from the atmosphere followed by an eventual emission occurs. Thus, to account for temporary carbon storage we use an alternative characterization method for GHG emissions, the Time Adjusted Warming Potential (TAWP) (Kendall 2012). The TAWP uses the relative cumulative radiative forcing (CRF) between an emission or removal of a GHG at a particular point in time (within the analytical time horizon of 100 years) and an emission of CO_2 today, resulting in units of CO_2 equivalents (CO_2e) *today*. The total kg ha^{-1} in CO_2e added per year in almond, walnut, and pistachio tree biomass was estimated using the logistic biomass model described above and in (Kendall et al. 2012), along with the total biogenic emissions from lost biomass at orchard clearing in order to calculate the TAWP of CO_2e drawn out of the atmosphere in tree growth. Storage in prunings, hulls, and shells is not considered in this calculation, as they are assumed to be mulched or otherwise disposed of in the same year that they are produced, resulting in a negligible contribution to carbon storage.

2.7. Scenarios

Two scenarios were examined: a typical or "business-as-usual" (BaU) scenario in which the best estimates of current co-product management practices and biomass fates was analyzed; and a maximum energy production scenario in which all available orchard biomass waste was utilized for energy production, including prunings, dead trees, shells, hulls, and cleared biomass at the end of the productive lifespan. Orchard clearing biomass was assumed to be directed to solid fuel power plants throughout the Central Valley as is the current practice, and prunings, shells, hulls, and individual removed trees were assumed to be combusted in modular biomass gasification-pyrolysis systems such as the Biomax system produced by Community Power Corporation (Overend 2004). Detailed discussion of the assumptions entailed in these two scenarios applied to almond production is offered in (Kendall et al. 2012), and the same assumptions applied for walnut and pistachio analysis in this study.

3. Results

Analyzed over a 61 year time horizon for a business-as-usual scenario, we find that the mean annual net GHG emissions of almond, walnut, and pistachio production are 1771, 2247, and 2986 kg CO_2e ha^{-1} respectively. These results include co-product credits of -1165, -159.8, and -137.9 kg CO_2e ha^{-1} and temporary storage credits of -187.4, -981.3, and -356.4 kg CO_2e ha^{-1} for almond, walnut, and pistachio respectively (Figure 5). Expressed per kilogram yield almond, walnut, and pistachio are responsible for 1.88, 0.89, and 2.17 kg CO_2e kg^{-1} kernel respectively (Figure 6).

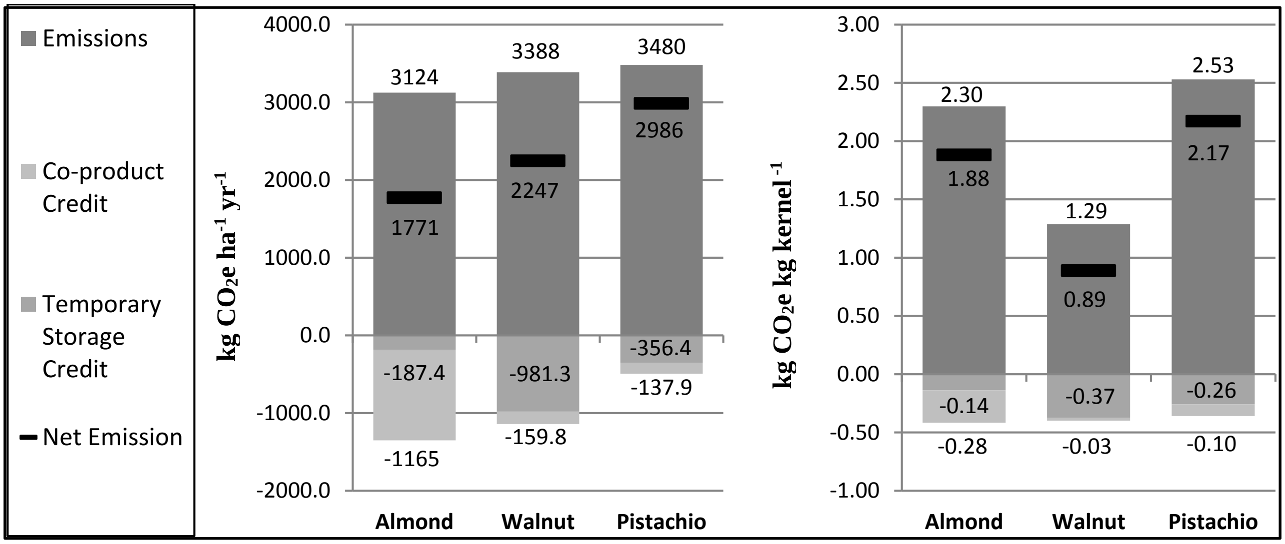

Figure 4. Mean annual GHG emissions and credits for nut orchards per hectare and per kilogram kernel under "business-as-usual" scenario.

The distribution of emissions among various management categories is roughly similar between almond, walnut, and pistachio, and is dominated by nutrient management at 38-43% total GHG emissions and irrigation at 16-34% total GHG emissions.

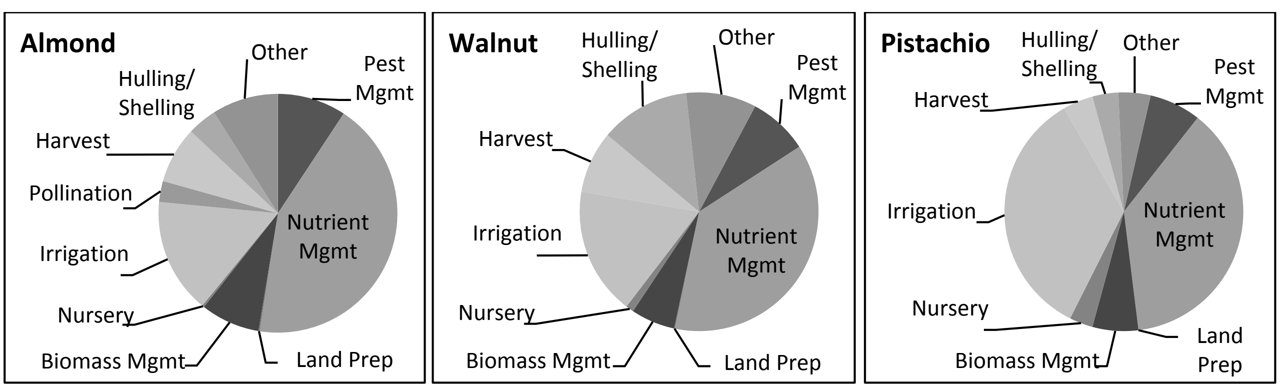

Figure 5. Breakdown of orchard emissions by management category for almond, walnut and pistachio.

The hypothetical maximum energy production scenario resulted in a substantial increase in potential co-product credits and a corresponding decrease in net emissions. Under this scenario net GHG emissions were calculated as -195.2, -981.3, and -84.67 kg CO_2e ha^{-1} and -0.14, -0.16, and -0.06 kg CO_2e kg^{-1} kernel for almond, walnut, and pistachio respectively (Figure 6).

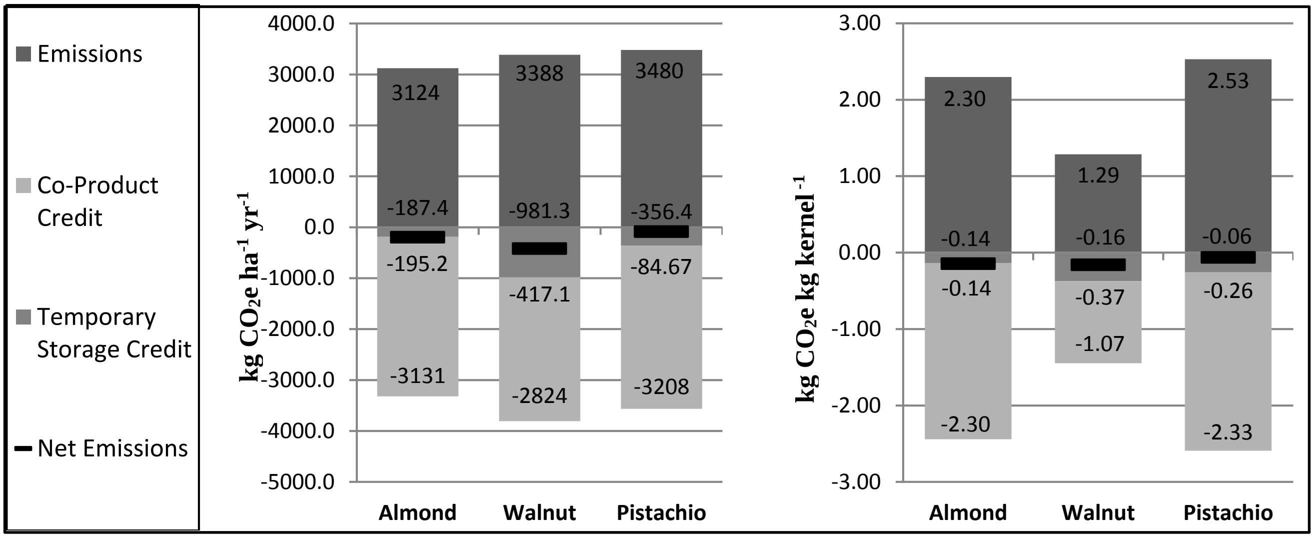

Figure 6. Mean annual GHG emissions and credits for nut orchards per hectare and per kilogram kernel under “maximum energy” scenario.

4. Discussion

The mean annual GHG burden, normalized to a 61 year time horizon, is similar among almond, walnut, and pistachio (Figure 4). This is due to the similarities in orchard management practices and to some extent, balances among inputs (Figure 5). For example, in wetter regions more frequent weed control and fungicide application is required, but less water is needed, whereas the opposite is true in dryer regions (Micke & Kester 1997). Somewhat greater differences in GHG footprint are observed when considering emissions per kilogram kernel (Figure 4) and when examining potential co-product and carbon storage credits. The former is due to variations in total yield and yield component ratios between these three crops (Table 2) while the latter is due to differences in co-product utilization (i.e., use of almond hulls as dairy cattle feed) and biomass accumulation characteristics (Figure 3). The scenario analysis highlights the variations between crops in potential to increase GHG credits through management practice – in particular, the expanded use of hull, shell, and waste biomass for energy production can generate significant GHG offsets (Figure 6).

5. Conclusion

This analysis examined typical almond, walnut and pistachio production in California using weighted-average data and consensus values for production inputs. As with all agricultural products, these nuts are subject to the inherent variability of region and climate which affects yields, biogeochemical emissions from orchard soils, and cultural practices of growers. This analysis highlights the critical importance of understanding the fate of co-products from orchard production including their utilization for energy production in order to determine true net GHG footprints of production, as well as the importance of maintaining high yields for increased GHG emissions “efficiency” of production (i.e., kg CO_2e kg^{-1} economic product).

The examination of a maximum energy scenario revealed the extremely high potential for California nut orchard systems to act as a net reducer of atmospheric GHG concentrations with the adoption of increased biomass co-product utilization practices and technologies. Indeed, due to lack of data, this analysis does not even consider the potential for carbon sequestration in orchard floor soils, which has been estimated as potentially highly significant (Kroodsma & Field 2006), nor does it consider the potential for increased management for biomass production (e.g., increased planting density coupled with periodic thinning, increased pruning frequency, cover cropping and no-till management, etc). Similarly, no scenarios accounting for potential GHG emission reduction through changes in inputs, management, or efficiency at the orchard field and production levels were examined. These represent additional potential sources for greenhouse gas reduction credits that will be explored in future analyses.

The results of this analysis suggest that combining the potential biomass management practices considered here with reductions in input intensity and alternative, low-GHG and energy intensive management practices could potentially allow California nut production to act as a significant GHG offset for the state. In order to promote and incentivize these practices, California's current GHG cap-and-trade legislation (Pavley & Nunez 2006) must properly credit orchard growers with the current beneficial results of perennial crop production, account for both co-product utilization and temporary carbon storage in standing biomass in calculation of net orchard emissions, and actively promote management practices that maximize the GHG benefits of these factors. Further research is needed to refine the walnut and pistachio LCA models and include more regionally specific data, as well as to elucidate the specifics of economic incentives that would be required to achieve optimum adoption of the GHG reducing management practices highlighted in this study.

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Environmental impacts of milk production in southern Belgium: estimation for nine commercial farms and investigation of mitigation via better manure application

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ABSTRACT

Cattle milk production is strongly criticized for its impacts on the environment. For the first time an LCA approach was used to estimate them in southern Belgium. Based on a 9-farms survey, mean (SD) environmental impacts of 1 kg of fat-and-protein-corrected milk were estimated as 8.4 (4.5) g PO_4eq, 15.5 (2.8) g SO_2eq, 1.4 (0.2) kg CO_2eq, 3.0 (2.9) CTUe, 3.8 (1.2) MJ and 4.6 (0.7) m^2.yr for eutrophication, acidification, climate change, ecotoxicity, cumulative energy demand, and land occupation, respectively. Improved practices for manure application to land according to UNECE (2007) were simulated for 7 farms as an option to mitigate direct NH_3 emissions, with emphasis on mineral fertilization and machinery use. Maximum reductions of 10 and 13% were estimated for eutrophication and acidification respectively, with only marginal effects (from -1.3% to 4.2%) on the other impact categories. Inter-farm differences in impact per kg of protein produced and per m^2.yr of land occupation indicates that there are potential improvement possibilities to be investigated.

Keywords: milk, mitigation, farm gate, life cycle assessment

1. Introduction

Production of ruminant-based foods has been strongly criticized for its contribution to damaging emissions to the environment and resource use (Steinfeld et al. 2006). However at a global scale, it is expected that demand for these products will continue to increase (Alexandratos and Bruinsma 2012). Since resources are limited and damaging emissions have to be controlled, there is a need to improve cattle production systems. In Wallonia (southern Belgium), milk production is an important agricultural activity. Indeed, it accounts for 24% of the added value produced in Walloon agriculture (SPW-DAEA 2011). Walloon milk production tends to be intensive; specialized milk farms have a mean apparent N surplus of 189 kg/ha/y (CRA-W 2011).

Since agricultural activities are involved in most major environmental problems in Wallonia (e.g. acidification, surface water pollution, greenhouse gas (GHG) emissions; KIEW 2013), and to avoid transferring impacts between impact categories in mitigation proposals, a multi-criteria approach was used to estimate the environmental pressure exerted by dairy systems in southern Wallonia. Even though GHG emissions per unit of milk produced in Belgium are among the lowest in the world (Gerber et al. 2011), large variations are expected at a regional scale (Casey and Holden 2006), and it seems worthwhile to explore this variation to find mitigation opportunities at the farm level. Furthermore, the link between milk production, regional production potential, and sensitivity at the territory level has to be considered to optimize land occupation (e.g. use of non ploughable land for food production) and to avoid exceeding limits of local pressure on the environment.

Until now, multiple environmental impacts of milk production in southern Wallonia have not been estimated with life cycle assessment (LCA) methodology. LCA was applied using a tool called "Weden", adapted from van der Werf et al. (2009), to 9 commercial farms located in a small territory (Gaume), and individual mitigation options based on improved manure application were proposed.

2. Methods

2.1 Goal and scope

The purpose of this work was to estimate environmental impacts of milk production by dairy farms in the Gaume region using an approach similar to that described by van der Werf et al. (2009). It is a one-year attributional cradle-to-farm-gate LCA that estimates environmental impacts of producing milk. The functional unit was 1 kg of fat-and-protein-corrected milk (FPCM; IDF 2010). The environmental impact categories considered were climate change (excluding land use and land-use change; IPCC 2006), eutrophication, acidification (CML 2013), cumulative energy demand (Hischier et al. 2010), ecotoxicity (USEtox 2013) and total land occupation (Hischier et al. 2010). System boundaries included main inputs and farm processes up to the farm gate, except for buildings and veterinary or cleaning products (Fig. 1). Waste disposal, destruction, or recycling were excluded. It was considered that imported animals and manure had no environmental burden but they were included in nutrient balance calculation. For all farms, classic technical indicators, such as N surplus and milk production per cow were also calculated to investigate improvement possibilities and their relation to environmental impacts. The target public was (1) farmers, to whom LCA results were reported along with technical indicators to investigate mitigation options mainly based on better manure application to land and (2) farm advisors, to consider LCA approaches for benchmarking and mitigating environmental impacts of cattle-based products at a small-territory scale (Gaume).

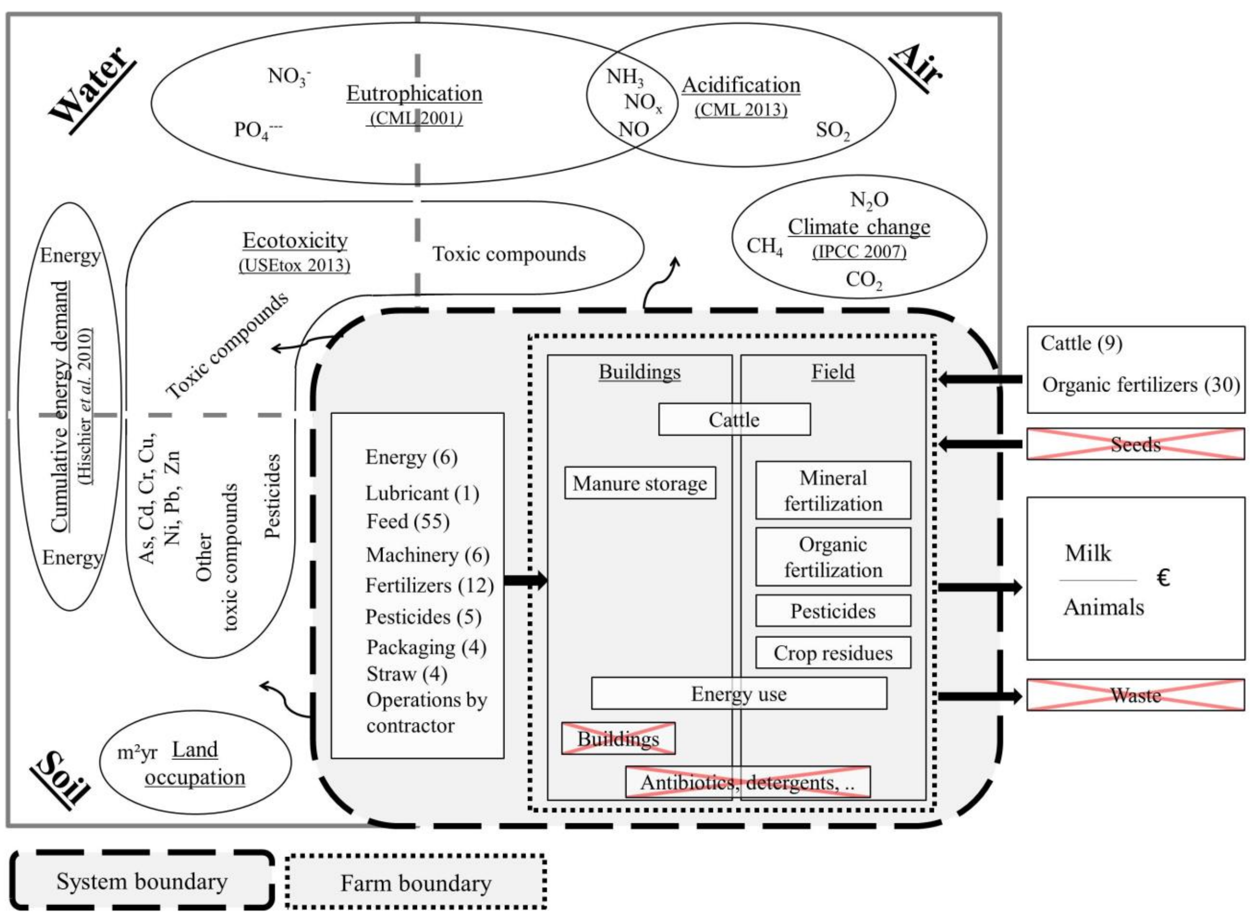

Figure 1. System description, impact categories and related damaging molecules and resources considered. Numbers in parentheses indicate the number of items considered in the inventory after aggregation (e.g., 63 machinery alternatives recorded on farms were aggregated into 6 groups).

2.2. Dairy production in Gaume, farm descriptions and technical indicators

The Gaume territory is a cultural sub-region that covers two-thirds of the Jurassic region of Belgium and includes 372 km² of usable agricultural area (UAA). It was chosen for this study because of its homogeneous soil and climate conditions, production practices, and cultural characteristics (to allow sociological analysis of production systems in the Qualaiter project). In 2010, in the Jurassic region 7271 dairy cows on 132 farms (25% of all farms) produced a mean of 5500 l of milk sold/cow/yr (CBL 2013). Only 39 of these farms were specialized dairy farms without a suckling herd. Despite the fact that mixed systems are dominant in this region, specialized dairy farms were mainly chosen to avoid problems of allocation or boundaries between dairy and beef or crop production in mixed farms. Nine farmers agreed to participate in this study, and 8 of them were exclusively dairy farmers (two of them organic), while the ninth had 32 suckler cows along with 120 dairy cows. Overall, the 9 farms held 895 dairy cows or 19% of dairy cows in the Gaume territory (Table 3). For the only mixed system, information was collected from the farmer to separate both systems. When no data were available to segregate crop- and animal-production systems (e.g. agricultural area used as pasture) an allocation between systems was done according to animal-feed dry-matter requirements. Data were collected on farms for the 2012 calendar year.

2.3. The Weden tool, inventory and impact assessment

The Weden tool consists of an Excel® spreadsheet split into three sections: input data, calculations, and reporting. Weden estimates emissions from the following farm processes:

- cattle: CH_4 emissions from enteric fermentation
- feed: emissions from production and importation of feed and straw
- induced emissions: N_2O emissions due to on-farm NH_3, NO_x and NO_3^- emissions
- machinery: emissions for machinery production
- mineral fertilizers off- and on-farm: emissions from production and importation and emissions from on-farm fertilization, respectively
- energy off- and on-farm: emissions from production and importation of energy and lubricants and emissions from on-farm energy and lubricant use, respectively
- organic fertilizers on-farm: emissions in the barn, during storage and at application
- pesticides and plastics off- and on-farm: emissions from production and importation of pesticides and plastics and on-farm emissions of pesticides to land, respectively
- others: on-farm emissions from crop residues and surpluses of trace metals, NO_3^- and PO_4^{---}

Table 1. Identification and explanation of modules/data sources of the IPCC and EMEP models adapted.

Domain	Adaptation	Variable used	Direct influence on
Herd management	Herd distribution among 4 periods: in barn, always on pasture (excluding milking time), intermediate periods (in barn at night, outside during the day) before and after winter	Duration of periods, duration of presence in the barn for milking and intermediate periods	Manure distribution, animal energy and protein requirements, animal diet
N fixation	Estimated according to legume species in grasslands	Mean proportion of legume species in grassland cover in June	N balance
Animal	Diet quality for each period and each animal category	Feed supplied (amount and quality[1]) and grazed grass quality	N and methane emissions
	Performances and characteristics	Weights, liveweight gains, proportion of pregnant cow, carcass yield, milk production, lactation period	Nutrient balances, cattle requirements, meat production, milk production
	Milk-production dynamics	Main periods of calving and milking duration	Dairy cow energy and protein requirements during the 4 periods of the year considered
Buildings	Barn type per animal category	Type of barn	Manure type and nutrient content
Manure management	Type of machinery and time before incorporation into soil	Machine type and duration before incorporation into soil	Ammonia emissions

[1] Digestibility and crude protein content

Models used for the on-farm inventory are based on element balances (N, P, K and trace metals) at the farm level (van der Werf et al. 2009). Methane (CH_4) and nitrous oxide (N_2O) emissions are modeled according to IPCC (2006) Tier 2 except for soil N_2O (Tier 1). Ammonia (NH_3), and nitrogen oxide (NO_2 and NO_x) emissions were calculated with EMEP/EEA (2009) equations. Phosphorus emissions to water were estimated according to Nemecek and Kägi (2007) while the nitrogen surplus at farm level was considered as lost to water as nitrate. Inventories were based on farm data collected from bookkeeping, other official documents related to the EU Nitrate Directive (AGW 2011) or common agricultural policy and farmer interviews (ca. 3 hours long) for farm management (cattle diet, barn type, manure management, etc.). This was done to increase model sensitivity to farm management and/or characteristics (Table 1).

The cattle module was divided into the ten animal categories used for national statistics and farm bookkeeping based on animal age and function (dairy vs. suckler cow). For each, animal requirements and excretion were calculated from IPCC Tier 2 (2006) based on diet quality, animal characteristics and management. If these were not available, data were taken from default values found in the literature (Table 2). Three manure types were considered: slurry, semi-solid manure and farmyard manure. Emission factors from IPCC (2006) used for semi-solid manure were those of solid storage and those for farmyard manure were those of storage below animals, according to emissions observed by Mathot et al. (2013). The Weden tool was also developed to take into account regional variability in machinery, barn types, manure management, pesticides, crop yield and characteristics, and organic and mineral fertilization (Table 2). Based on a detailed list of 279 machines (Mecacost tool: Rabier et al. 2008), details given by farmers about annual use (converted to kg machine equivalent), and the corresponding inventory, emissions of 6 categories of machines (general, tillage, harvester, tanks, tractors and trailers; Frischknecht et al. (2013)) were estimated. Consumption of fuel for operations by contractors was estimated according to Rabier et al. (2008) and expert opinion (F. Rabier, Pers.comm. 2013) for the choice of representative equipment and use. Inventories for the production of pesticides (5 types), mineral fertilizers (12 types), energy sources (6 types), lubricant (1 type), and plastics (4 types) were obtained from Frischknecht et al. (2013) and detailed data from farmer interviews and regional product lists and characteristics when necessary (Table 2). Inventories of feed and straw production were adapted mainly from Blonk et al. (2009), Frischknecht et al. (2013) and Piazzalunga et al. (2012).

Table 2. Main data sources for inventory and modeling of emissions

Item	Primary sources
Nutrient contents of slurry, semi-solid manure, farmyard manure and types of organic matter	AGW 2011; Godden et al. 2013;
Nitrogen fixation by legumes in grasslands	Limbourg et al. 2001
Machinery characteristics and fuel consumption	Rabier et al. 2008
Trace metal balance and content	Li et al. 2005; Piazzalunga et al. 2012
Cattle characteristics	INRA, 2007
Plant yield and characteristics; feed composition	CVB 2010; INRA 2007
Animal product characteristics	Mathot et al. 2012;
Mineral fertilizer characteristics	Piazzalunga et al. 2012
Pesticides	SPW-DAEA 2012
Packaging	van der Werf et al. 2009

Emission factors for eutrophication (PO_4eq), acidification (SO_2eq) and climate change (CO_2eq) were identical to those used by van der Werf et al. (2009). Cumulative energy demand (MJ) was calculated using characterization factors of Frischknecht et al. (2007). For ecotoxicity (CTUe), emissions of all toxic compounds included in the ecoinvent v2.2 inventory database (Frischknecht et al. 2007) were included for inputs, using USEtox characterization factors (USEtox 2013). On-farm emissions of trace metals, products of combustion and active ingredients of pesticides were included. A simplified approach was used for active ingredients of pesticides: we assumed that all active ingredients applied were emitted to soil. A list of currently applied products (467) and their active-ingredient concentrations (239) was obtained from SPW-DAEA (2012). When available (ca. 75% of pesticides), characterization factors from the USEtox database were used (not shown). When not known, the mean characterization factor of all active ingredients was used (486 CTUe/kg).

Per farm, one mitigation option to decrease NH_3 emission was simulated taking into account a better manure management by application of slurry with trailing hose instead of a splash plate and rapid incorporation of solid manure (UNECE 2007). A change in spreading equipment (tractor and tank/spreader) was included based on Rabier et al. (2008). Regardless of the mitigation option, and according to simulation with the Mecacost tool (Rabier et al. 2008), fuel consumption did not increase due to the use of more efficient spreading equipment (suitable power and faster operations). Mineral N fertilization was also reduced by assuming a ratio of 1 kg N per kg of N-NH_3 not lost during organic fertilizer spreading.

2.4 Uncertainty analysis and interpretation of results

Sensitivity to the type of allocation method used for co-products (meat and milk, no plant production was sold) was tested by comparing results from economic, biophysical (IDF 2010) and protein allocation. The latter two allocation methods were calculated using the change in animal numbers after subtracting imported animals, while for economic allocation, neither changes in animal numbers or purchases were included. Statistics were performed with R software (R Development Core Team 2011). For each farm, a summary of the results was produced, including comparison with those of other farms. The summary included results about the technical indicators, impact on the environment per kg of FPCM and per m^2 of UAA, and distribution of environmental impacts between on- and off-farm activities and within the farm. Furthermore, opportunities for mitigation were discussed based on manure management while using more efficient machinery to avoid NH_3 losses at spreading. Partial results and technical indicators (Table 3) were used to validate data collected and results. To investigate links between production efficiency and land-use intensity, the relation between impacts per m^2 of land occupation and impacts per kg of total protein produced was analyzed. Total protein was used as indicator to include meat production.

3. Results

Using economic allocation, mean (standard deviation (SD)) emissions per kg of FPCM for the 9 farms were 254 (116) g CO_2, 1.04 (0.2) g N_2O, 8.3 (1.71) g NH_3, 2.27 (0.38) g NO, 0.88 (0.36) g NO_x, 0.54 (0.2) g SO_2 and 32 (6) g CH_4 to air and 0.01 (0) g NH_3, 39 (39) g NO_3^-, 1.11 (0.83) g PO_4^{---} to water. Due to the large number of active ingredients involved in ecotoxicity, it was not possible to disaggregate them. Mean environmental impacts for the production of 1 kg of FPCM were 8.5 (4.5) g PO_4eq, 15.5 (2.8) g SO_2eq, 1.4 (0.2) kg CO_2eq, 3.0 (2.9) CTUe, 3.8 (1.2) MJ and 4.6 (0.7) m².y for eutrophication, acidification, climate change, ecotoxicity, CED and land occupation, respectively (Fig 2.1-2.6), with economic allocation between milk and meat production. High variability was observed for all impact categories (coefficient of variation = ca. 10-100 %, Fig. 2.1-2.6). Changing the allocation method to biophysical or protein-based allocation (Table 3) reduced impacts of milk production by 11% and 4%, respectively. However, there were no relations between economic and biophysical or protein allocations ($p>0.05$, $r=0.225$ and $r=0.178$, respectively), though protein and biophysical allocations were significantly correlated ($p<0.001$, $r= 0.997$).

For eutrophication, acidification, climate change and land occupation, on-farm emissions dominated (>70%), while for CED and ecotoxicity off-farm emissions dominated (>60%). Organic matter management (storage and application) was, on average, a main contributor to eutrophication (46% (30%)), acidification (84% (7%)) and climate change (22% (3%)). However, the main contributor to climate change was the herd, through CH_4 emissions from enteric fermentation (48% (8%)). For ecotoxicity, feed production contributed 65% (20%), while pesticide production and application contributed 9% (10%).

The main source of N fertilizer was organic manure. Indeed, 51.4% (17%), 8.6% (13%), 5.2% (10%) of N fertilized originated from slurry, farmyard manure and semi-solid manure, respectively, while the remaining 35% (26%) came from mineral fertilizers. On average, 87% (6%) of NH_3 emissions were due to organic manure management, with 34% (6%), 30% (4%) and 36% (10%) of them were emitted in the barn, during storage and spreading on the soil, respectively. Considering the equipment for and management of solid and liquid manure

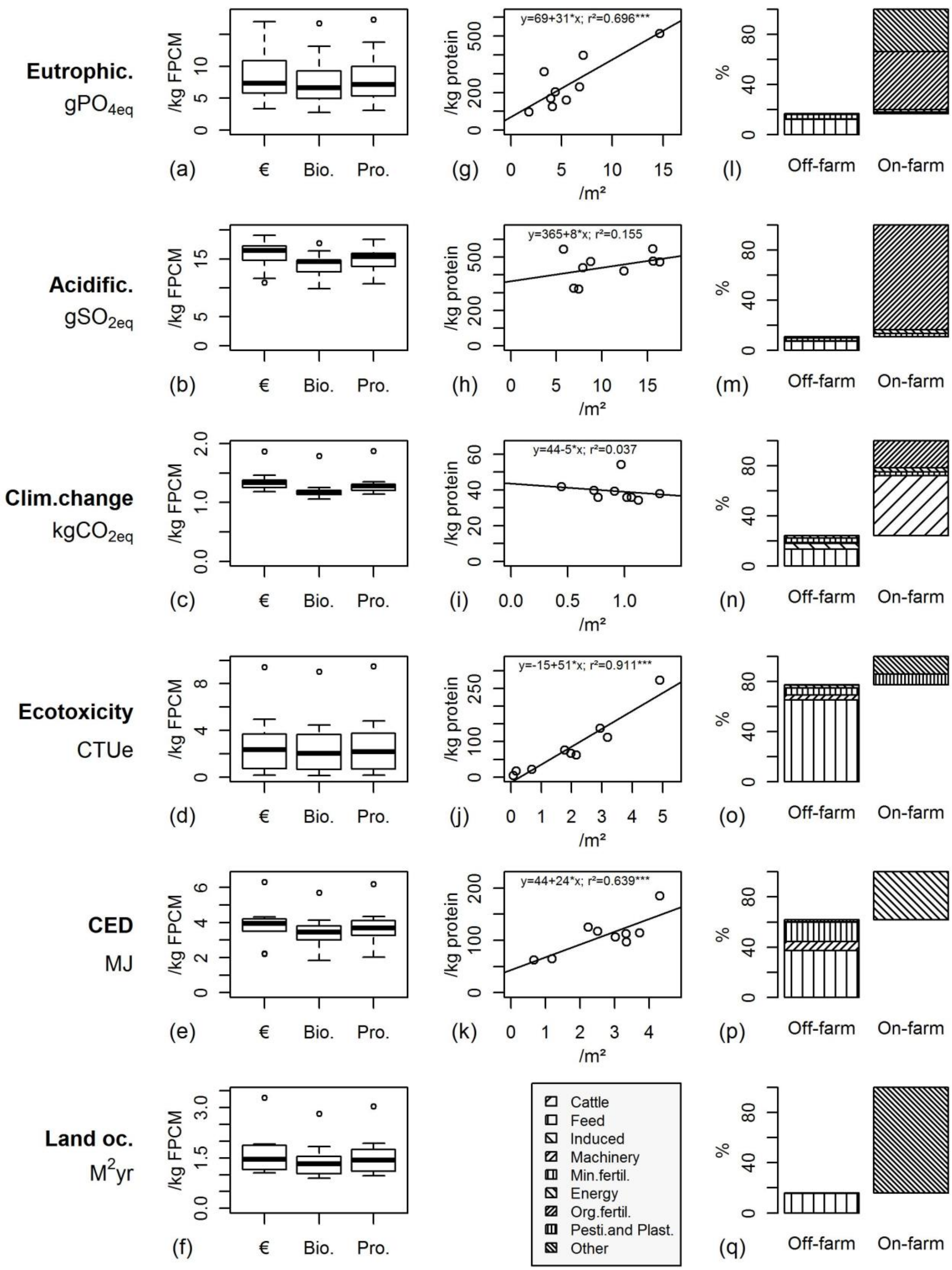

Figure 2. Potential environmental impacts per kg of fat-and-protein corrected milk (FPCM) as a function of allocation method (2.1-2.6): economic (€), biophysical (Bio.) and protein (Pro.), boxplots according default parameters. Relations between impact per m^2 of land occupation and that per kg protein produced for the 9 farms (2.7-2.11); *,** and *** indicate $p<0.05$, $p<0.01$, and $p<0.001$, respectively. Distribution (2.13-2.17) of the environ application to land, there was an opportunity to mitigate of NH_3 emissions and consequently N fertilizer use for

7 farms (Table 3). Simulation of these mitigations led to a decrease of 26% (6%) of NH_3 emission at application and, consequently, with a change in equipment and reduction of mineral fertilization, to reductions of 4.8% (3.6%) and 8.4% (2.6%) of eutrophication and acidification impact, respectively. Marginal positive effects were also observed for the other impact categories (climate change: 0.4% of CO_2eq, ecotoxicity: 6% CTUE and CED: 0.7% of MJ), but an increase was observed for the farms without mineral N fertilization (Table 3).mental burden: off- (left) and on-farm (right) and, within each, origin of the impact.

For eutrophication, ecotoxicity and CED, significant positive correlations were observed between impacts per kg of protein produced and per m².y used ($p<0.05$). No correlation was observed ($p>0.05$) for the other impact categories (Fig. 2.7-2.11).

Table 3. Mean characteristics, technical indicators, allocation factors and results of mitigation options for all 9 farms studied

Category	Item	Unit	Mean	SD	Max	Min
On-farm area	Total	Ha	94	41	159	55
	Grasslands	%	73	14	96	53
	Silage maize	%	17	15	35	0
	Other crops	%	10	8	27	0
Herd	Total	Number	197	62	334	117
	Dairy cows	Number	99	37	175	42
	Head not dairy cows/head dairy cows		1.05	0.30	1.80	0.74
	First calving age	Month	31	5	36	25
	Livestock units (LU)[1]	LU/ha/year	2.4	0.6	3.2	1.2
Milk production	Total	10^3 kg milk	733	365	1559	299
	Without milk for veals	10^3 kg milk	688	352	1493	270
	Per dairy cow per year	kg milk	7461	1309	9188	5373
	Production duration/calving interval	%	84	3	89	80
	Fat concentration	g/kg milk	40.1	1.8	43.2	37.5
	Protein concentration	g/kg milk	33.4	1.6	35.2	31.0
Meat production	Live weight produced	10^3 kg	20.0	6.2	27.8	11.2
	Protein produced	10^3 kg	3.00	0.93	4.17	1.68
Balance[2]	N surplus	g N/m²	14.1	8.7	33.4	5.0
	N fixation	g N/m²	2.97	1.67	5.89	1.05
	N inputs in feed and fertilizers	g N/m²	57	31	116	20
	N fertilization[3]	g N/m²	17	10	31	4
	NH_3 losses during manure fertilization	g N/m²	1.92	1.12	3.69	0.69
	P surplus	g P/m²	1.60	1.57	3.92	-0.98
	P inputs in feed and fertilizers	g P/m²	2.78	1.48	5.03	0.80
Impact allocated to milk	Economic	% of € from milk/ € revenue	91	2	95	88
	Protein-based	% of kg protein sold as milk /kg protein produced	88	3	93	82
	Biophysical		82	5	89	73
Mitigation[4]	NH_3 emissions	% N lost as NH_3	26	6	33	17
	Eutrophication	%	4.8	3.6	9.7	1.0
	Acidification	%	8.4	2.6	12.5	4.5
	Climate change	%	0.4	0.3	0.8	-0.3
	Ecotoxicity	%	0.6	1.7	4.2	-1.2
	Cumulative energy demand	%	0.7	1.3	1.8	-2.0
	Land occupation	%	0	0	0	0

[1] According to Stilmant et al. (1998). [2] At the farm scale, related to the usable agricultural area of the farm. [3] Organic and mineral fertilization at spreading (before NH_3 losses from manure). [4] Percentage reduction for the 7 farms considered for mitigation.

4. Discussion

Although comparing results of this study with those from the literature is difficult because of methodological differences, we estimated higher values than those reported by Nguyen et al. (2013) for eutrophication (+60%) and acidification (+33%), similar values for climate change (-4%) and land occupation (+10%) but smaller values for CED (-14%). For eutrophication and acidification, differences could be explained in part (<10%) by high NH_3 emissions due to the low efficiency of equipment used in 7 of the farms. For eutrophication, 260 (300) g PO_4eq came from NO_3^{--} emissions of 14.1 (8.7) g N/m², higher than those in a similar study (8.1 g N/m²; van der Werf et al. 2009), which could help to explain the high eutrophication impact estimated. However, too little information was available for a full comparison with Nguyen et al. (2013), who did not calculate NO_3 emissions using farm-gate N balances. Furthermore, our small sample size means that extreme values can greatly influence mean values. When reporting results to farmers, environmental impact, technical information and mitigation possibilities (Table 3) were discussed with a particular attention to extreme values, like for nitrogen surplus, to investigate other mitigation options than those concerning manure management. As no environmental burden was attributed to imported animals, impacts are underestimated. However, for the 9 farms considered animals were imported on only one farm (2% of the herd), thus the underestimation is certainly relatively small.

The allocation method also influenced impacts of milk production, attributing fewer impacts to milk when shifting from economic to protein-based to biophysical allocation. Attributional LCA remains problematic due price fluctuations impact on economic allocation or more generally the choice of allocation methods that may change distribution of impacts between meat and milk. For product labeling, attributional approach seems necessary (e.g. IDF 2010); however, the use of functional units that require no allocation between co-products at the farm level, such as protein production (van Dooren et al. 2014), could be used to assess improvement in food production. Consequential LCA or system expansion could also be used.

Considering total protein production (milk and meat), improvement may be achieved by adopting the mitigation option of improving manure application to land or management. Simulation of the former, even though based on simple models, decreased environmental impacts (by 0.4-8.3%). The simulation assumed a reduction of N mineral fertilization directly proportional to the decrease in NH_3 emissions. This assumption is open to criticisms in the short term at least due to farmers' fears of quality or yield losses and higher costs due to change in machinery or energy purchase. Since NH_3 emissions from organic fertilization strongly influence eutrophication and acidification impacts, higher reductions could be expected if more efficient equipment was used (e.g. manure injection; EMEP/EEA 2009). This possibility was not tested because it would have involved major changes in equipment and energy use, reducing the chances that farmers would adopt the mitigation option. Furthermore, no reduction in mineral fertilization was possible in organic systems, which do not use it. Thus, only changes in machinery were taken into account, leading to marginally higher emissions for other impact categories (negative values in Table 3). This problem reflects difficulties in predicting changes in the quantity and quality of potentially available N for plant production and the related milk production, which was not predicted in simulations.

For mitigation associated with changes in farm management, even though only a few farms were considered, the positive relation between impact per kg FCPM and per m^2 of land occupation estimated for eutrophication, ecotoxicity and CED should be investigated further. Indeed, this relation indicates that it is feasible to have relatively low impact per unit of product and per unit of land occupied. Indirect emissions contributed strongly to ecotoxicity and CED, while direct emissions (N and P surpluses) contributed strongly to eutrophication (Fig 2.12, 2.15 and 2.16). Therefore, the relation between feed inputs and animal production (milk and meat) seems to be a key factor to adjust to reduce environmental impacts. However, on-farm land occupation is quite high (0.842 (0.098) m²/total m²), reflecting the importance of on-farm cattle feed production and of milk production linked to the territory, which is composed mainly of grasslands (0.731 (0.142) m²/m² of on-farm land occupation). Other mitigation options can be proposed based on technical indicators such milk production, herd composition or age at first calving. The complexity of production systems at the farm scale, however, makes it difficult to render good advice about practices. For example, to reduce the ratio of non-productive to productive animals, which varies widely between farms (coefficient of variation = 30%, Table 3), one could recommend reducing the age at first calving. However, one consequence of this change could be an increase in the amount of concentrated feed in the diet of young cattle, which could increase environmental impacts but also decrease veal production. Less veal production would have potential consequences, such as, in a consequential LCA approach, increased veal production in other systems to satisfy demand for veal.

Finally, the variability observed highlights the diversity and thus room for improvement in these production systems, even in a small and relatively homogeneous area. Methodological problems, however, such as errors in the inventory are still possible due to missing or incorrect data collection or modeling. Therefore, validating results and/or improving models are necessary. For example, in this study the complete causal chain recommended by IPCC (2007) and EMEP/EEA (2009) for cattle feeding was included, from feed ingestion to manure application to soil. However, the causal chain for N emissions probably depends greatly on the quality of the diet used in the inventory. Ideally, feed input, feed stock change and, when available, forage analysis in diets included in the study have to be validated. Results such as mean N (excretion – losses in the barn and storage) and CH_4 production by cattle (105 (21) kg N and 130 (13) kg CH_4 per dairy cow per year, respectively) have to be compared to regional recognized referential values. Furthermore, in the future, default values in causal chains should be replaced with better data, for example, by improving knowledge about cattle nutrition or estimating enteric CH_4 emissions at the farm scale with estimation based, for example, on milk analysis (Dehareng et al. 2012). However, this last approach will not solve the problem of modeling GHG emissions of non-milk-productive animals, which requires additional measurement (Mathot et al. 2012). Finally, beyond the normative approach, improvement and mitigation strategies have to be identified.

5. Conclusion

Despite the small number of farms studied, environmental impacts of milk production in the small territory of Gaume was highly variable. Implementing good practices such as using better manure spreading methods may mitigate eutrophication and acidification impacts. Further research is required to investigate and model differences in farm inputs and production, including both milk and meat co-products.

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Uncertainty analysis of cattle-based product LCA related to model variables: case study of milk production in Belgium

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ABSTRACT

Uncertainties in environmental impacts of milk production related to model variables were investigated with Monte-Carlo simulation in a case study. Per kg of fat-and-protein-corrected milk produced, the 95% confidence interval of impacts was 6.2-10.4 g PO_4eq, 10.1-25.6 g SO_2eq, 1.1-1.9 kg CO_2eq, -7.6-19.2 CTUe, 4.3-4.9 MJ and 1.11-1.28 m²yr for eutrophication, acidification, climate change, ecotoxicity, CED and land occupation respectively. Expressed as coefficients of variation, uncertainties ranged from 3% to 2097% as a function of the impact category. The most influential variables changed with impact category, except those related to the functional unit. Monte-Carlo simulation and sensitivity analysis help to identify variables requiring more accuracy and detect errors implementing multiple-variables models in calculation tools.

Keywords: distribution, Monte-Carlo simulation, life cycle assessment, IPCC

1. Introduction

Uncertainty assessment in life cycle assessment (LCA; ISO 2006) of agricultural products, milk in particular, is a major concern, and relatively few studies have considered it (Yan et al. 2011). Flysjö et al. (2011) showed the importance of uncertainty in some emission factors, such as N_2O from soil used in relatively simple models, on the uncertainty in the carbon footprint of milk production. The influence of emission factors on nitrogen (N) compound emissions at the farm level compared to calculation with the N balance has also been highlighted (Payraudeau et al. 2007). Since agricultural activities are involved in most major environmental problems in Wallonia (e.g. acidification, surface water pollution, greenhouse gas (GHG) emissions; KIEW 2013), Belgium, a multiple environmental impact approach is necessary for milk production. To this end, a tool called “Weden” based on van der Werf et al. (2009) was built that included the ability to explore uncertainty in environmental impacts. This study aimed to investigate uncertainties in environmental impacts of milk production related to model variables and identification of the most influencing variables with a Monte-Carlo simulation approach based on a case study.

2. Methods

2.1. Weden tool

LCA of milk production on a farm was performed with a tool called Weden, consisting of an Excel® spreadsheet, described in Mathot et al. (2014). Models used for the on-farm inventory were based mainly on element balances (N, P, K and trace metals) at the farm level. Methane (CH_4) and nitrous oxide (N_2O) emissions were modeled with IPCC (2006), mainly Tier 2. Ammonia (NH_3) and N oxide (NO_2 and NO_x) emissions were calculated according to EMEP/EEA (2009), mainly Tier 2, approaches. For these two models, causality chains for calculation of N and CH_4 emissions from cattle feed ingestion to manure spreading were fully implemented. Phosphorus emissions to water were estimated according to Nemecek and Kägi (2007), and inputs at the farm level were adapted mainly from Frischknecht et al. (2007) using SimaPro software (PRé Consultants 2007). System boundaries, compound targets in the inventory and environmental impacts considered are summarized in Fig. 1. In the model used it is considered that imported animals and manures had no environmental burden but they are included in nutrient balance calculation. However, in this case study no manure or animal was imported into the farm.

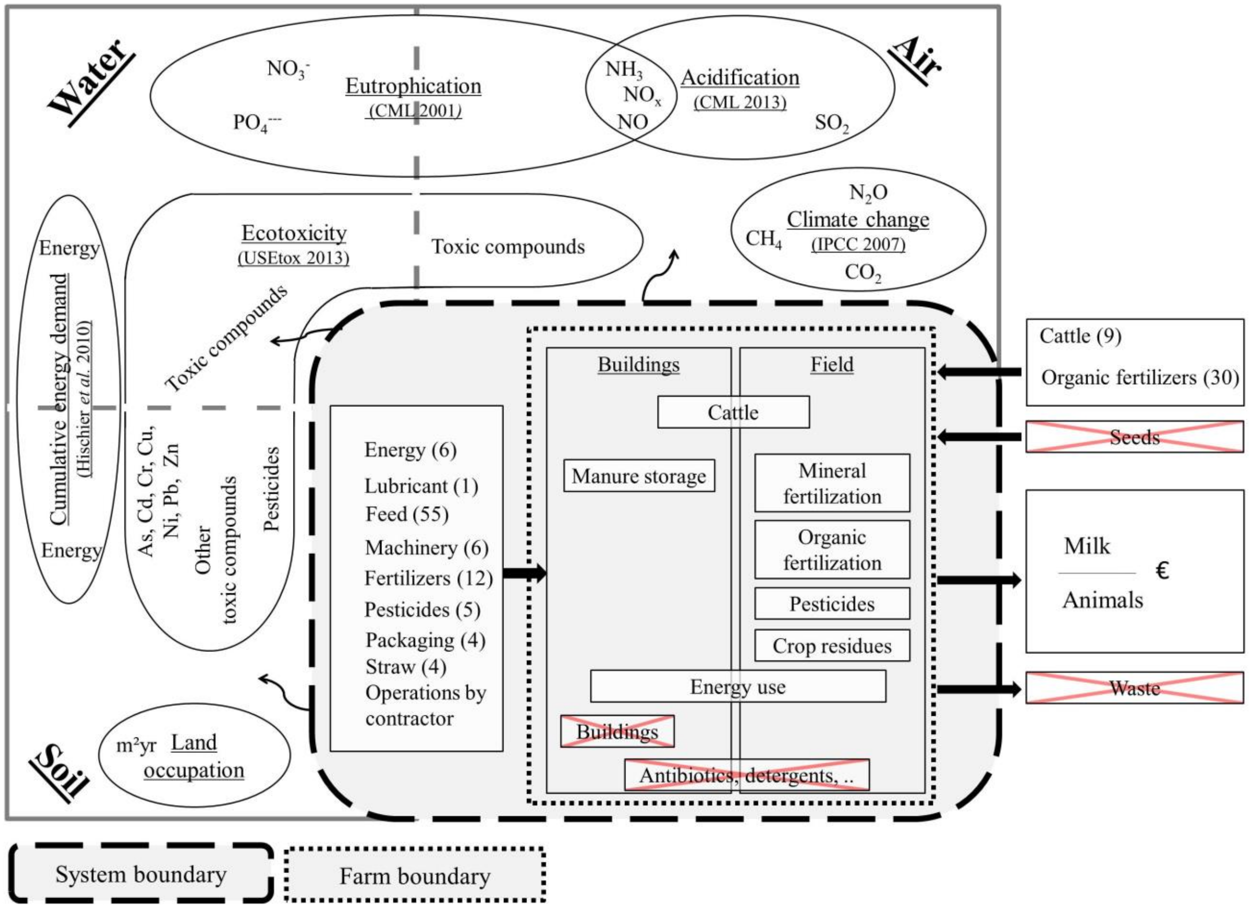

Figure 1. System description, impact categories and related damaging compounds and resources considered. Numbers in parentheses indicate the number of items considered in the inventory after aggregation (e.g., 63 machinery alternatives recorded on farms were aggregated into 6 groups).

2.2. Uncertainty calculation

Uncertainty analysis was performed with Monte-Carlo simulations and sensitivity analysis using three types of distribution: triangular, lognormal (with geometric mean) and normal (Weidema et al., 2013). Input variables supplied by farmer (Table 1) were excluded while model variables (e.g. mean cattle weight, N_2O emission factors, manure composition, feed N concentration) were varied in the uncertainty analysis. For each variable, a distribution was chosen based on characterization from available databases or literature data and a realistic range (e.g. nutrient concentration in slurry). When a variable's distribution was unknown, it was subjectively defined as normal with a coefficient of variation of 5% around the mean. Uncertainty in impact characterization factors was estimated only for ecotoxicity factors missing in USEtox (2013). Values for active ingredients used as plant protection products (PPPs) were considered to be lognormally distributed based on the geometric mean and standard deviation (SD) of the other PPPs in USEtox (2013) used. For some variables, Monte-Carlo simulation with not-well-characterized uncertainty parameters, as found in the literature, may lead to unrealistic variable values. For example, for dry matter content in certain manures, uncertainty parameters were arithmetic means and SDs, suggesting a normal distribution of the content. However, during Monte-Carlo simulation, it was possible to obtain negative values. If that occurred, the distribution was not changed; instead, the simulation was looped to consider only realistic minimum and maximum values. Monte-Carlo simulation was iterated 1000 times. A sensitivity analysis was also performed to investigate influence of variables with default uncertainty parameters and to identify major influencing variables. This operation consisted of recording changes in all impacts associated with a change in a single variable equal to the 75th percentile of its distribution. Two results were recorded: relative change in the impact, calculated as $Var=(Y_{0.75}-Y_{0,5})/Y_{0,5}$, and relative change in the impact divided by relative change in the variable, $Sen=Var/((X_{0.75}-X_{0,5})/X_{0,5})$, where Y is the impact value and X is the variable value. Indices indicate the percentile in the distribution corresponding to the value used.

Var indicates the proportion of change in the impact according to the change in the variable at its 75th-percentile value. It allows ranking variables according to their influence on results but relies strongly on the uncertainty parameters chosen for a given variable; therefore, it is not completely suitable for variables with default uncertainty parameters. Indeed, for these variables, strong underestimation or overestimation can be suspected. Sen indicates the relative sensitivity of the impact to the change in the variable. It is useful for identifying variables with default parameters whose precision has to be increased due to their potentially large influence on impacts.

Table 1: Input variables not considered in the uncertainty analysis.

Domain	Input variable
Cattle management	Head of cattle (in, out, losses, change in stock); milk production, consumption of veal, protein and fat concentrations; proportion of pregnant cows; lactation and non-lactation duration; distribution of calving period; grazing duration; diet composition
Crop management	Surface areas; amount and type of mineral and organic fertilizations; plant protection product amount and type used
Manure management	Amount produced and exchanged (in and out); type; treatment; application system
Revenue	From milk, from meat, from crops
Input	Amount of feed, straw, fertilizers, plant protection products, energy, plastics
Machinery	Type; amount and use
Operations by contractors	Type; amount

2.3. The case study

One of the farms analyzed by Mathot et al. (2014) was chosen as case study because of the supposed reliability of its input variables. This farm contained 77 dairy cows, for a total dairy herd of 168 head of cattle. Usable agricultural area was 67.4 ha, with 80% covered by grasslands, 12% by silage maize and 7% by other crops. Total milk produced in the year investigated, 2012, was 669.5 10^3 liters of milk, with a mean concentration of 42.4 g of fat and 33.5 g of protein per kg of milk produced. Inputs through feed and fertilization were 63.7 g N/m² and 1.43 g P/m². Of total revenue, 93.7% came from milk, and 6.3% came from meat.

3. Results

3.1. Distribution

According to the Shapiro normality test (R Development Core Team 2011) only the predicted values for CED were normally distributed. Per kg of fat-and-protein-corrected milk (FPCM) produced, impacts (95% confidence intervals) were 6.2-10.4 g PO_4eq, 10.1-25.6 g SO_2eq, 1.1-1.9 kg CO_2eq, -7.6-19.2 CTUe, 4.3-4.9 MJ and 1.11-1.28 m²yr for eutrophication, acidification, climate change, ecotoxicity, cumulative energy demand (CED) and land occupation respectively. Impact means (and CV) were, respectively, 8.2 g PO_4eq (13%), 16.3 (25%) g SO_2eq, 1.4 (14%) kg CO_2eq, 10.0 (2097%) CTUe, 4.6 (3%) MJ and 1.2 m²yr (3%) (Fig. 2).

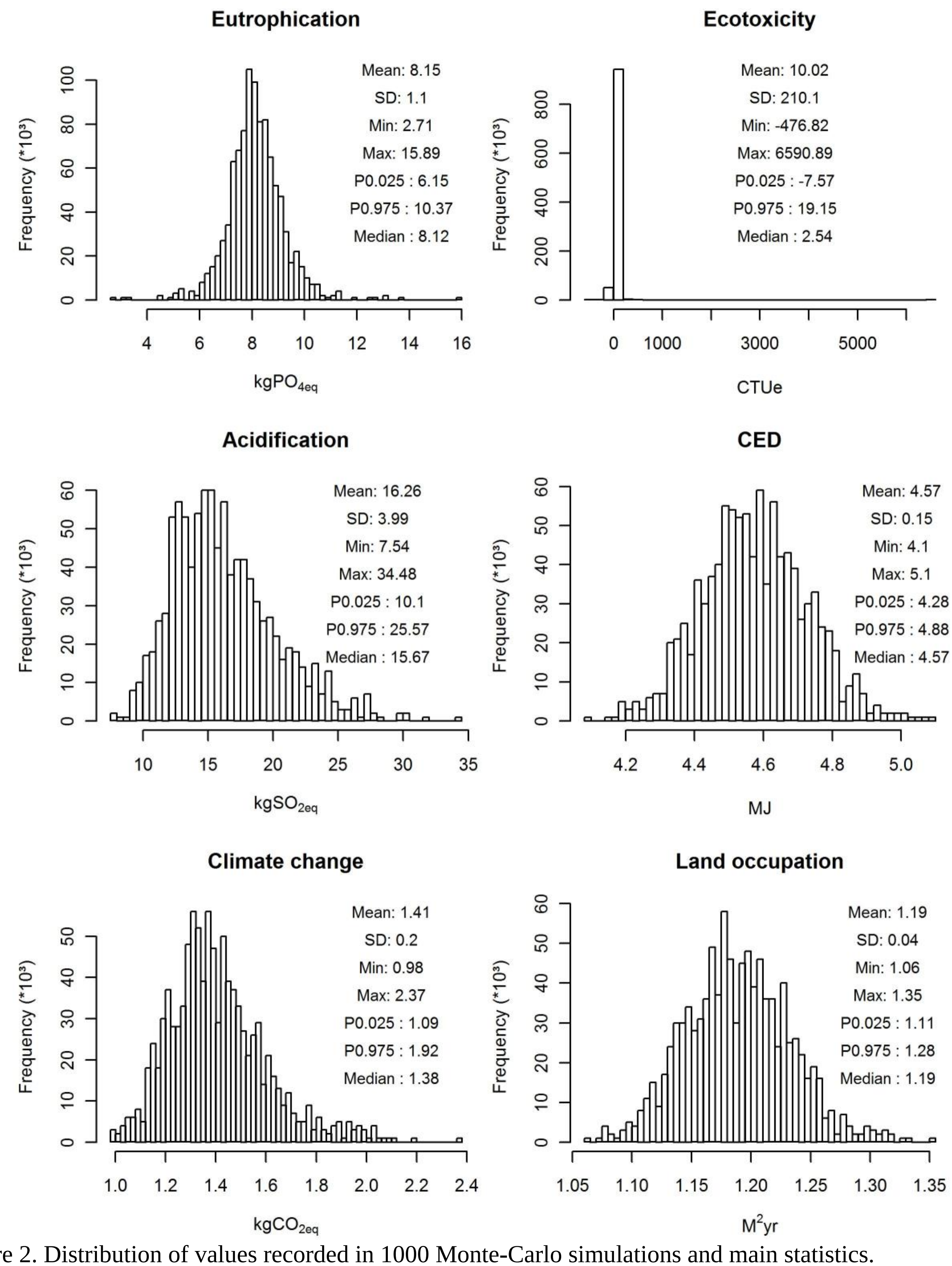

Figure 2. Distribution of values recorded in 1000 Monte-Carlo simulations and main statistics.

3.2. Variation and sensitivity

In total, 1084 variables influenced the results, all impact categories included. For 85% of the variables, the uncertainty in parameters was taken from literature, calculated or estimated. For the remaining 15%, uncertainty parameters were default values.

Table 2. Number (n) of variables influencing environmental impacts. The variable n indicates the number of variables influencing Var and Sen at more than |±1%| and |±10%|, respectively. Min and max are the values of the minimum and maximum influences observed. Moy<0 and Moy>0 are the means of negative and positive influences, respectively.

		Var						Sen					
		N		Value				n		Value			
Impact	n	+1%	-1%	Min	Moy <0	Max	Moy >0	+10%	-10%	Min	Moy <0	Max	Moy >0
Eutro.	314	3	1	-0.038	-0.001	0.048	0.001	5	7	-0.496	-0.024	0.402	0.011
Acidif.	221	8	10	-0.064	-0.006	0.088	0.003	14	23	-1.908	-0.175	2.528	0.060
Climate ch.	278	6	8	-0.041	-0.003	0.055	0.001	13	16	-1.222	-0.092	1.631	0.029
Ecotoxicity	141	2	7	-0.031	-0.004	0.042	0.002	4	3	-0.496	-0.070	0.268	0.009
CED	59	1	0	-0.017	-0.008	0.005	0.001	4	5	-0.496	-0.310	0.244	0.026
Land occ.	48	1	0	-0.017	-0.008	0.009	0.001	4	0	-0.496	-0.310	0.063	0.004

Table 3. Classification (number) of variables influencing the impacts more than |±1%| for Var and more than |±10%| for Sen. "Default parameters" is the number of variables using default parameters

	Eutrophication		Acidification		Climate change		Ecotoxicity		CED		Land occupation	
Variable	Var	Sen	Var	Sen	Var	Sen	Var	Sen	Var	Sen	Var	Sen
Feed and manure composition	0	0	4	11	2	5	1	0	0	0	0	0
Other input characteristics	1	1	0	0	0	0	3	0	0	4	0	0
Output characteristics	0	0	0	0	0	0	3	0	0	0	0	0
Fat-and-protein-corrected milk	1	3	1	3	1	3	1	3	1	3	1	3
Animal diet requirement	0	2	7	17	5	12	0	0	0	0	0	0
Emission factor	2	0	4	1	4	2	1	3	0	0	0	0
Other modeling	0	2	0	1	0	1	0	0	0	0	0	0
Conversion factor	0	4	2	4	2	6	0	1	0	2	0	1
Default parameters	1	8	14	33	10	26	1	3	1	3	1	3

3.2.1. Relative changes in impacts

CED and land occupation were influenced by relatively few variables (<60) while others were influenced by many more (141-341; Table 2). Eutrophication, acidification, climate change and ecotoxicity were subject to larger changes (greater than |±2%|) from the most influential variables than CED or land occupation (less than |±2%|). Regardless of the impact category, less than 10% of the variables changed the impact more than |±1%|.

3.2.2. Relative sensitivity of impacts

Climate change and acidification impacts were highly sensitive to certain variables. For more than 10 variables, the sensitivity was higher than |±10%|. Furthermore, these two impacts were highly sensitive to some variables (>|±100%|).

3.2.3. Influential variables

The most influential variables (>|±1%| for Var and |>±10%| for Sen) were identified and classified in 8 categories: Feed and manure composition; Other input characteristics (e.g. fuel CO_2 emissions on combustion); Output characteristics (e.g. protein content in animal live weight sold); FCPM (equation for calculation of functional unit); Animal diet requirement (model for estimation of cattle requirement and manure production), Emission factors (e.g.: N_2O emission from fertilization applied to soil); Other modeling (e.g., estimation of N fixation by plants) and Conversion factors (e.g., protein:N ratio of milk). For Var and Sen, the numbers of major (>|±1%| for Var and |±10%| for Sen) influencing variables were 4 and 12 for eutrophication, 18 and 37 for acidification, 14 and 29 for climate change, 9 and 7 for ecotoxicity, 1 and 9 for CED and 1 and 4 for land occupation, respectively (Tables 2 and 3). Among them, 1 and 8 for eutrophication, 14 and 33 for acidification, 10 and 35 for climate change, 1 and 3 for ecotoxicity, 1 and 3 for CED and 1 and 3 for land occupation were characterized with default parameters for Var and Sen, respectively. As expected, for all impact categories, conversion of milk production to FPCM influences the results, with the standardization of milk fat content as main contributor. Typically, results are sensitive to conversion factors but do not vary much because of them due to their low uncertainties, except for dry-matter-to-energy conversion and protein-to-N conversion, for which default parameters of distribution were attributed. Feed and manure composition and animal diet requirement variables influenced acidification and climate change impacts, while eutrophication impact was mainly influenced by emission factors in the model of N fixation by legumes. Ecotoxicity and CED were mainly influenced by other input characteristics and emission factors.

4. Discussion

Uncertainties in multiple environmental impacts of milk production were explored by considering mainly inventory model variables. Uncertainties, expressed as coefficients of variation, ranged from 3-2097% as a function of the impact category. The most influential variables changed with the impact category, except those used to calculate the functional unit. For this calculation, the most influential variable was milk fat content, which implies that accurate knowledge of milk fat content is necessary. Acidification and climate change impacts were influenced by many variables related to cattle requirements and feed composition. This reflects the impact of using IPCC and EMEP models, which consider the causal chain from feed ingestion to manure application and its input data through diet characteristics (digestible energy and protein content). When using these models, it is important to have good knowledge of diet composition. The influence of emission factors emphasizes the effect of a few generic values (e.g. N_2O emissions from soils). CED and ecotoxicity impacts were calculated with balance models; consequently, they were mainly influenced by input-output characteristics or emission factors attributed to input production. This is also partially the case for eutrophication, which is based on balance equations for some compounds, such as on-farm nitrate emission, but also on emission models, such as on-farm phosphate emission.

In this study, the influence of uncertainty in input variables was not included but would certainly add large uncertainty to the results. This work highlights the influence that different variables have on different impact categories, showing that impact can be sensitive to variables whose uncertainty is high or not well characterized. The procedure used to calculate uncertainty has some limits, however, such as not taking into account covariance between the variables of the same model. A global model approach would be interesting to improve the understanding of uncertainty. In the future, uncertainty related to the complexity of estimating emissions of damaging compounds should also be considered using model comparison and validation. Ultimately, performing Monte-Carlo simulations, as in this study, may help identify errors in implementation of models in calculation tools.

5. Conclusion

This approach helps to bound uncertainties related to model variables, which can be quite high, in environmental impacts of milk production. In the future, the identification of the most influential variables in emission models should help in decreasing uncertainties by improvement of their accuracy. Finally research is required to investigate influence of input variables.

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Why Practice Life Cycle Management? Case Study of the New Zealand Wine Industry

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ABSTRACT

The NZ wine sector is one of the most active NZ agribusiness sectors in terms of environmental sustainability initiatives and yet individual wineries vary widely in their commitment to environmental sustainability. The aim of this exploratory research, therefore, was to analyse why NZ wineries respond to the environmental sustainability agenda and to the Life Cycle Management (LCM) dimensions of this agenda. The results of this study indicate that adoption of a life cycle perspective in a certification programme is not a "selling point" for companies considering certification; however, the life cycle approach can lead to identification of new insights and related initiatives. Those advocating for adoption of LCM may wish to highlight these benefits as opposed to – or in addition to – expounding a narrative around expanded environmental responsibility along the supply chain.

Keywords: Life Cycle Management, New Zealand wine industry, environmental certification

1. Introduction

Environmental sustainability is increasingly relevant in the marketplace, and can provide a point of competitive advantage for companies that differentiate themselves on the basis of environmental credentials. However, there are marked differences between companies (as well as between economic sectors) in integration of environmental aspects into business activities.

New Zealand is a fast growing competitor on the New World wine map, and the New Zealand wine industry has recorded a strong expansion in both wine production and exports in the last decade (NZ Winegrowers 2013). The NZ wine sector is one of the most active NZ agribusiness sectors in terms of environmental sustainability initiatives and yet individual wineries vary widely in their commitment to environmental sustainability. The aim of this explorative research, therefore, was to analyse how and why (some) NZ wineries respond to the environmental sustainability agenda and to the Life Cycle Management dimensions of this agenda.

A prototype conceptual framework identifying the potential key factors explaining the attitudes and activities of companies regarding environmental sustainability was used to guide the research (see Section 2). Based on this framework, face-to-face interviews were undertaken with a small sample of NZ wineries (see Section 3). The results were then analysed against the framework (Section 4), and discussed in relation to the application of Life Cycle Management (Section 5).

2. Conceptual Framework

The need to explore linkages between business culture and sustainability is encouraging companies to reconsider their operations and strategies (Hall and Howe 2012).

Business culture is concerned with how involved parties (manufacturers, distributors, suppliers and retailers, government and shareholders) conduct their business (Bhaskaran and Sukumaran 2007; LaBahn and Harich 1997). Business culture is affected by many different external and internal factors including: social culture factors (religion, education, beliefs and attitudes, etc.); political, economic and technological factors; trade environment and globalisation; and business factors (size, ownership, management, etc.). The relationship between business culture and social culture, dynamic external factors, and business factors has been studied by a number of authors (e.g. Hofstede 1994; Wallace et al. 1999; Fang 2003; Fletcher and Fang 2006; Singh 2007). However, limited research has been undertaken on how business culture, shaped by different external (national and industry bodies, market and economic environment) and internal factors (company structure, management and marketing) affect a company's commitment to environmental sustainability (Packalen 2010). A prototype conceptual framework describing this relationship is shown diagrammatically in Figure 1.

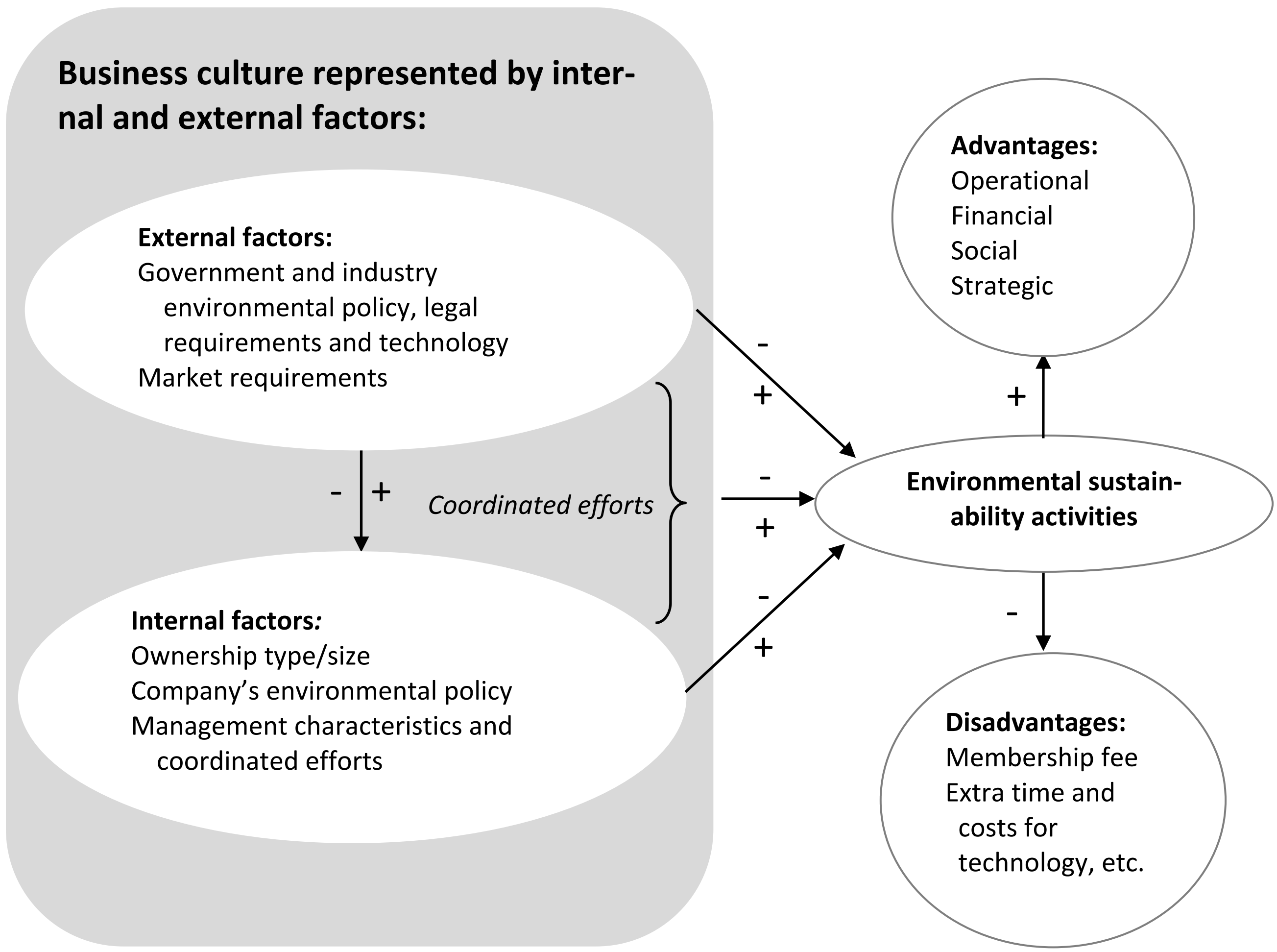

Figure 1. Conceptual framework describing relationships between business culture and environmental sustainability activities in companies.

In this study, environmental sustainability refers to the impacts of a company's activities on the natural (or modified natural) environment and to how these are managed. The environmental sustainability agenda comprises the issues that are considered relevant for management such as encouraging biodiversity, reducing greenhouse gas (GHG) emissions, and decreasing use of synthetic pesticides. This exploratory research analysed the influence of a number of specific external and internal factors/determinants on pro-active operations, management and strategies for environmental sustainability in NZ wine sector.

3. Methods

A qualitative methodology was employed in this study using semi-structured interviews. Ten New Zealand wineries were included in the study using snowball sampling. The interviewees were either the general/environmental managers or the owners of the wineries. All of the wineries included in this research had adopted some type of environmental sustainability certification. Seven wineries were members of the Sustainable Winegrowing New Zealand (SWNZ) programme, four had carboNZero/CEMARS certification, and three had organic BioGro certification. All interviews used open-ended questions and were recorded.

Qualitative data analysis was used for this research including the iterative process of describing, classifying and connection. Ethics issues were a high concern in this research relating to the participants' privacy and confidentiality.

4. Results

4.1. Summary of the NZ environmental sustainability programmes

Three key environmental sustainability certification programmes were used by the NZ wineries: Sustainable Winegrowing New Zealand (SWNZ), carboNZero (or CEMARS), and BioGro (organic certification scheme). In addition, one winery had ISO14001 certification and another one had had this certification in the past, and one winery had the UK's Carbon Trust's Carbon Footprint Label. Only the carboNZero and the Carbon Trust Carbon Footprint Label programmes use a Life Cycle Assessment (LCA) based approach i.e. they assess the greenhouse gas emissions associated with products along their full life cycles from extraction of raw materials, through manufacture, distribution, use and on to final waste management.

SWNZ (Sustainable Winegrowing New Zealand) was established in 1995 as an industry initiative, and was adopted by growers from all the grape growing regions in 1997. The SWNZ programme provides a framework for viticultural and winemaking practices. The seven focus areas in the SWNZ programme are: biodiversity; soil, water and air; energy; chemicals; byproducts; people; and business practices. Certified companies submit a report every year and are independently audited at least once every three years. Certified companies may put the logo on their products.

Established in 2001, **carboNZero** is an internationally accredited greenhouse gas certification programme. There are five key steps to attaining certification through the carboNZero programme: measure, manage, mitigate, verify, and market. The carboNZero programme may certify organisations, products, services, events, or individuals – and the carboNZero logo may be used on certified company websites, products and other publicity. Companies produce an annual report disclosing their greenhouse gas emissions and activities taken to reduce their carbon footprint, and are regularly audited by an independent auditor. CEMARS (Certified Emissions Measurement And Reduction Scheme) certification is an alternative certification option for organisations, also offered by the carboNZero programme, which focuses on the measurement and manage steps of the carboNZero programme.

BioGro is a New Zealand-based organic certification system that certifies producers, food processors, distributors and others to organic standards. Certified organisations are independently audited every year, and certified companies may put the BioGro logo on their products. The three organic wineries studied in this research also practiced biodynamic production techniques but had not opted for biodynamic certification.

4.2. External factors influencing environmental sustainability initiatives

Analysing the activities of the wineries against the external factors summarized in the conceptual framework (see Section 2), it was clear that "government and industry group environmental policy, legal requirements and technology" was a key driver for environmental sustainability activities for most of the wineries studied. In particular, in 2007 NZ Winegrowers launched its Sustainability Policy which stipulated that wines from vintage 2010 on must have been produced under one of a small number of recognised, independently audited, sustainability programmes in order to be included in NZ Winegrowers' national and international marketing, promotional and awards events. Effectively this meant that almost all NZ wine producers moved to achieve either SWNZ certification or organic/biodynamic certification by the end of 2012 (NZ Winegrowers, 2014a, 2014b).

Regarding market requirements, the interviewees did not identify any specific environmental requirements being demanded by their customers. At its most pronounced, the interviewees described "picking up signals" in their markets. For example, one interviewee said, *"Every time I went and did a wine tasting overseas, I'd always be asked how many sprays we use, do we use sprays in the vineyard ... So that was one reason for us to look more closely at what we were doing."* It also seemed that the economic environment (vis a vis the global economic crisis) had not made a significant difference to the way the wineries operated in their international markets with respect to responding (or not) to any change in demand for environmental sustainability credentials. However, a couple of the interviewees did mention that the international popularisation of the Food Miles concept in 2006/7 (see McLaren 2009) had created pressure for them to respond to that issue.

4.3. Internal factors influencing environmental sustainability initiatives

With respect to internal factors summarised in the conceptual framework, the type of ownership and/or size was not correlated with environmental sustainability activities: proactive environmental management was just as likely to be practised by a family-owned winery as by a business partnership, and by the smallest as by the biggest wineries. The existence of a company environmental policy/vision also appeared to be uncorrelated with environmental sustainability activities; in fact, only one of the wineries appeared to have formal procedures in place for monitoring and directing environmental sustainability performance (as far as this could be distinguished from the specific requirements of any of the certification programmes). On the other hand, "management characteristics and coordinated efforts" seemed to be important in determining the direction and scale of environmental sustainability activities. Particularly where a company had been founded by one or more current owner-managers, the environmental sustainability activities reflected their own beliefs; for example, one interviewee explained*, "The reason we had a lot of successes in that," referring to sustainability activities, "is because [X, the company founder] introduced it from before the winery was built."*

Regarding coordination amongst staff, differing perspectives were apparent. Some managers expressed frustration that staff did not spontaneously support environmental initiatives; for example, one remarked*, "... until you get your own vineyard managers and workers mindset into what they see, it is very hard for me to push it. And just say, right go and do this. Because you've got to have them on board as well."* However, in another winery many of the environmental initiatives were initiated, trialled and developed by staff throughout the company; the interviewee explained the context for this buy-in to environmental sustainability from staff:

> *"So, we are not a company that's been going for ten years and saying, "OK guys, from tomorrow we are going to have a sustainability focus," because everyone would think, "Oh yeah, that's another management thing they've come up with, it'll last a month and then be back to normal". So, whoever comes on board here know that that's part of what [X] is about. It's not that hard to keep that enthusiasm culture going, because that's just what we are."*

One interviewee identified the age of staff as a significant factor:

> *"I think it is the older people like me that have to think about it and think why are we doing this, or this is a waste of time separating all these out or doing this – and have to be convinced. I think the younger generation are much more inclined to just take it on just as part of what they are doing and they are the ones probably that come to us with more ideas about how we can change all the lights to energy saving lightbulbs or change the lights in rooms that we don't use very often so that they've got a sensor so that they come on and off as people come and go. But they are thinking about it just as a natural part of their life."*

4.4. Advantages of engaging in environmental sustainability initiatives

Turning to the perceived advantages and disadvantages of environmental sustainability activities, for most of the wineries the perceived advantages could be summarised as operational and strategic rather than financial or social. The biggest perceived advantage of environmental sustainability activities seemed to be the proactive strategic positioning of the company as indicated by this quote:

> *"If you were an importer and you were choosing between two or three different wineries, and the products were relatively the same, and the price was relatively the same, and someone was carbon neutral and if you had the option to put that on your brand, it is another thing that would lead them to maybe choose us ...*"

Regarding operational advantages, the main benefits expressed were related to the certification programmes and the need to regularly examine one's own processes as part of certification procedure (whether it was SWNZ, carboNZero or BioGro). As one interviewee remarked, *"It's a good way of especially, well in both the vineyard and the winery, of just keeping people accountable for what they do, making sure that record keeping is accurate and timely."* At another winery, the interviewee commented about the company's involvement in the carboNZero programme, "*Once we got all this measurement criteria, that helps us identify what projects we want to look at ... A lot of our projects we put in, to address the information we get from being part of carboNZero."*

Regarding financial benefits, the interviewees did not consider that they had received any price premiums for their wine products as a result of their activities. Furthermore, although the various certification processes had identified opportunities for more efficient energy use, freighting, etc., the cost savings were considered to be outweighed by the costs of the measurement and certification processes. Only three of the wineries were able to identify contracts where their environmental sustainability credentials had been a factor in winning the contracts; two of these wineries were only able to identify one contract each whilst the third winery (which was extremely proactive in publicising its sustainability credentials) considered that it was a (minor) factor in securing several contracts.

There was relatively little evidence of social advantages associated with environmental sustainability activities. Indeed, one interviewee mentioned how one member of staff complained about *"spending too much time and money on the gardens."* However, two interviewees did make the link between environmental initiatives and benefits for staff; as one said, "*I think it is good for the staff because if you can show them the benefits of it and get them behind it, it certainly makes a better work environment*."

For the organic wineries, the advantages of organic production concerned first and foremost the quality of the wine. As one interviewee responded when asked why their winery used organic practices, *"Because it improves the quality overall of the wine ... I know the quality is better …You can see the difference. If we couldn't see an improvement we wouldn't do it, we would go and do something else that gave us better quality. It is only done for quality reasons for us, it is not a marketing ploy."* For these wineries, organic or biodynamic production methods did not comprise a set of optional activities but were fundamental to the *raison d'être* of the wineries. Therefore a discussion about the advantages versus disadvantages of engaging in such activities was considered irrelevant.

4.5. Disadvantages of engaging in environmental sustainability initiatives

The interviewees did not identify any major disadvantages associated with engaging in environmental sustainability initiatives apart from the financial costs associated with the certification processes (charges for auditing, etc.) and staff time required to prepare certification documents. This is perhaps not surprising for the wineries certified to carboNZero given that they voluntarily chose to engage in that certification programme. However, for the SWNZ certification which is effectively compulsory (unless another approved scheme was chosen instead), several interviewees expressed the view that it was too easy *and "people looked at it and said you are not really telling me anything I don't know."* For both the carboNZero and SWNZ certification programmes, the interviewees considered that there should be more benchmarking so that they could identify how they were doing relative to the other wineries, and that the certification programmes should undertake more marketing to raise their profiles. The interviewees at the organic wineries certified to BioGro expressed frustration at the narrow focus of the carboNZero certification programme (i.e. the focus on carbon footprinting as opposed to a range of environmental impacts) and the lack of stretch in the SWNZ programme (i.e. they felt that the SWNZ programme could be more proactive in requiring wineries to demonstrate environmental improvements).

4.6. Interpretation from a Life Cycle Management perspective

Life Cycle Management (LCM) is a specific type of environmental management that focuses on managing the environmental impacts associated with products and services along their life cycles from extraction of raw materials, through manufacture, distribution, use and on to final waste management (Remmen et al., 2007). For wine products, obviously this includes managing the environmental impacts related to vineyard and winery activities as well as those associated with the upstream manufacture and delivery of inputs, and the downstream distribution and consumption of wine. The results in Sections 4.1 to 4.5 are therefore relevant to a discussion on uptake of LCM. However, in addition, it is worthwhile to consider evidence of, and attitudes to, activities that are particularly indicative of uptake of LCM. In this research, these activities were classified into three areas: (a) prioritisation of activities according to the magnitude of environmental impacts at different points in the cradle-to-grave life cycle of products and/or services; (b) vertical coordination with supply chain partners both upstream and downstream along the supply chain regarding environmental sustainability initiatives; and (c) changes to processes in order to close the loop for material and waste life cycles and to displace environmental impacts occurring at different life cycle stages.

Regarding (a), prioritisation of activities from a life cycle perspective, several Life Cycle Assessment (LCA) studies of wine have shown that the glass bottle makes one of, if not the, biggest contribution to a range of environmental impacts due to the energy-intensive glass manufacturing process and the weight of the packaging which must be transported from the glass bottle manufacturer to the winery and then on to markets (e.g. Amienyo et al. 2014; Barry 2011). Typical improvement activities include lightweighting of glass bottles, bulk shipment of wine prior to bottling, and use of alternative materials for wine containers (see, for example, Amienyo et al. 2014; Point et al. 2012; WRAP, 2007). All but one of the non-organic wineries mentioned lightweighting of their glass bottles as an environmental sustainability activity; however, this activity was not mentioned by any of the organic wineries. Only one winery had actually started to use plastic rather than glass bottles.

Other environmental hotspots are commonly identified at the vineyard production stage of the life cycle: toxicity impacts related to use of pesticides, and sometimes eutrophication and acidification impacts related to use of fertilisers (e.g. Barry 2011, p.20-21; Point et al. 2012). Activities in the vineyard that addressed these impacts were commonly mentioned in the interviews (e.g. planting wild flowers between vine rows, reducing use of pesticides). However, these actions focused on improvements and innovations within a single life cycle stage and therefore they did not constitute evidence of adoption of a life cycle perspective.

Regarding (b), vertical coordination along the supply chain, several interviewees mentioned discussions with suppliers about environmental initiatives, and in particular the glass bottle manufacturer regarding lightweighting of bottles. Regarding (c), closing the loop and displacing environmental impacts at different life cycle stages, this type of thinking was evident in the composting activities practised by all the wineries where waste organic matter from the winery was composted and applied out in the vineyards. However, only one winery had innovated to close the loop for other material and energy life cycles; at this winery, vine prunings were burned to generate heat that displaced use of liquid petroleum gas (LPG) in the winery, and a pilot project was underway to produce biochar from the vine prunings for application to the vineyard soils. In addition, this winery had also focused on displacing environmental impacts arising at different life cycle stages by, for example, installing wind turbines and solar panels, and grazing sheep in the vineyards (to displace the use of diesel for mowing).

5. Discussion and Conclusion

Perhaps the first point to note is that the organic wineries had a quite different motivation for engaging in environmental sustainability activities (assuming that organic practices are recognized as a subset of environmental sustainability practices). For them, it was necessary to be organic in order to produce wine of high quality. However, for the non-organic wineries, the key factors explaining their attitudes and activities regarding environmental sustainability were independent of the quality of the wine except in so far as quality was expressed through the brand story. One interviewee even hinted at a more definite distinction between organic and sustainable practices, saying, *"We are not organic. I think our main push is the sustainable thing."*

For the non-organic wineries, the factors that explained their attitudes and activities could be summarised as: a belief that environmental responsibility was "part of the story" about their products, that it would lead to more efficient business operations, and also that it placed them ahead of the competition in terms of environmental responsibility. For these wineries, they did not consider that they could achieve a price premium for demonstrating their environmental credentials; as one interviewee said, *"… for us it is a nice to have. It is part of what we do and we are not charging any more."*

Regarding enablers and barriers explaining adoption of environmental sustainability activities, external factors seemed to have a limited role apart from the industry sector requirement to be a member of SWNZ or another approved environmental certification programme. There was little evidence of demand for demonstration of environmental credentials in the marketplace, and instead the wineries regarded environmental sustainability initiatives as just a part of their brand story, and often not a particularly important part of that story. As one said *"No, it is not a strong part of our brand story at all."* However, it should be acknowledged the relative importance attached to this aspect did vary between the wineries. Only one internal factor appeared to be significant: the management characteristics and coordinated efforts.

Regarding the perceived advantages and disadvantages of engaging in environmental sustainability activities, the wineries considered that environmental sustainability activities enhanced their brand story, as noted above, and that the different certification schemes enabled them to "fine-tune" their operations. Some also considered that adoption of environmental sustainability activities created a better work environment for staff. The perceived disadvantages were related to the certification schemes rather than the environmental initiatives themselves; aspects that were mentioned included the costs of the schemes and the lack of awareness of the schemes amongst customers (which limited their usefulness as a marketing tool for the wineries).

Adoption of a life cycle perspective was evident in the decisions to undertake composting and to move to lightweight bottles; several of the wineries also mentioned discussions with suppliers regarding their environmental activities. However, only two of the wineries (which both had carboNZero certification) consciously articulated that they had taken a life cycle perspective in their environmental sustainability initiatives, and they both identified involvement in the carboNZero programme as a driver for new projects.

These results indicate that adoption of a life cycle perspective is not a “selling point” for companies considering environmental sustainability activities. Yet the life cycle approach can lead to identification of new insights and related initiatives with relatively significant environmental benefits in the life cycle of wine (such as a move to alternative packaging types). Such initiatives may assist the company in telling its brand story and attaining competitive advantage, being more efficient, and benefitting employees – factors that are recognised as advantageous by companies. Those advocating for adoption of Life Cycle Management may wish to highlight these benefits as opposed to – or in addition to – expounding a narrative around expanded environmental responsibility along the supply chain.

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Higher accuracy in N modeling makes a difference

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ABSTRACT

Simplified modeling of fertilizer N emissions in inventories of agricultural products may lead to considerable deviations of N emissions from actual N surpluses present in farming system. We compared impacts for global warming, eutrophication, and acidification of four crops either produced in organic or integrated (IP) production using ecoinvent inventories (v2.2). We remodeled N_2O, NO_3, and NH_3 emissions considering N input from fertilizers, N deposition, N fixation, management induced changes in the soil C-N-pool, and N withdrawal by yields. Global warming on a product basis of organic crops was up to 24% lower compared to IP after remodeling. Eutrophication of organic crops was comparable to IP crops and acidification initially up to 3.5 times higher in organic crops was only 1.9 times higher after remodeling. By taking total nitrogen turnover into account in the N modeling impacts of crops from different farming systems can be differentiated with higher accuracy.

Keywords: N emission modeling, fertilization, organic, conventional, total nitrogen turnover

1. Introduction

In the past 50 years agricultural intensification has led to a global increase in nitrogen fertilizer use by over 800%, which has led to a massive increase in food production (Foley et al., 2011). On a global scale food production and consumption contribute the highest share of reactive nitrogen compounds released to the environment (Leach *et al.*, 2012). This has environmental impacts on different levels: Nitrogen emissions contribute to global warming, eutrophication, acidification, and negatively impact biodiversity and human health (Galloway *et al.*, 2003). A more sustainable food production should therefore seek to reduce excessive fertilizer use and reduce nitrogen losses to the environment using a set of different strategies (Galloway *et al.*, 2008). One strategy often referred to in the scientific literature is the improvement of the N efficiency in animal and crop production (Galloway *et al.*, 2008; Foley *et al.*, 2011; Sutton *et al.*, 2011).

By prohibiting mineral nitrogen fertilization, organic agriculture is usually lower in its nitrogen intensity than conventional agriculture. In fact often in organic farming nitrogen is a yield limiting factor leading to lower N surpluses per ha. Analyses comparing N fluxes in different farming systems showed a higher N efficiency in organic crop rotations when taking into account the changes in soil organic nitrogen stocks (Küstermann *et al.*, 2010).

However, a recent review of comparative LCAs of organic and conventional products revealed often higher impacts for eutrophication, acidification and global warming for organic products (Meier *et al.*, 2014). A detailed analysis of several LCAs comparing organic with conventional products, though, showed that N emissions from fertilization as modelled in inventories may deviate considerably from the actual N-surplus present in the farming system. It was concluded that higher accuracy in N modeling would better differentiate between products from different farming systems.

A major problem in contemporary N emission modeling in LCA inventories of agricultural products seems to be that emissions models do not represent N emissions from organic fertilizers very well. This is particularly the case for soil born N_2O emissions and NO_3 leaching. Soil borne N_2O emissions from fertilizer input in LCA inventories are often modeled using the IPCC model (IPCC, 2006b), which estimates N_2O emissions based on the fertilizer N input only. By doing so the model considers the soil as black box not taking into account N fluxes in and out of the soil. However, especially for organic fertilizers where only a fraction of the total N is readily available for plants microbial processes within the soil are important for their mode of action (Gutser *et al.*, 2005) and in consequence determine their emission pattern. If high fractions of organic fertilizers are applied (as in organic farming were exclusively organic fertilizers are used) the amount of applied total ammoniacal nitrogen (TAN) is usually not enough to meet the crop plants' demand. The additional nitrogen needed for plant growth is supplied by readily available nitrogen mineralized from the soil C-N-pool. A simple N_2O emissions

model, which takes into account N mineralization from as well as N immobilization into the soil C-N-pool, has been proposed by Meier *et al.* (2012).
NO_3 leaching strongly depends on local climatic factors (rainfall), soil parameters and vegetation cover. Existing NO_3 emission models used in LCA inventories are usually too simplistic to cover the complex interactions between these factors. Accurate nitrate leaching requires the simulation of soil hydrological and biogeochemical processes (Li *et al.*, 2006). However, even with relatively complex models it is difficult to simulate nitrate dynamics accurate enough and a detailed parameterization is needed for satisfying results (Pedersen *et al.*, 2007). In part this is also due to the still poor understanding of the long-term fate of fertilizer-derived nitrogen in the plant-soil-water system (Sebilo *et al.*, 2013). Since NO_3 emissions from fertilization seem to be the most difficult ones to model an alternative solution is its calculation from the nitrogen balance by subtracting the modeled NH_3 and N_2O emissions from the N surplus. This in turn requires all N inputs and outputs such as fertilizer input, symbiotic N fixation, atmospheric N deposition, management induced changes in the soil C-N-pool and the N withdrawal by the yield. By doing so the law of mass conservation for N fertilization is satisfied, which is also the main requirement for calculating N use efficiencies (Godinot *et al.*, 2014).

The objectives of this study were to assess the chances in impact assessment for global warming, eutrophication, and acidification for four crops produced in different production systems when N emissions from fertilizers were remodeled on the inventory level using a N_2O model that also accounts for N turnover from the soil C-N-pool and adjusting NO_3 emissions to the actual N surplus.

2. Methods

In our analysis we considered the following ecoinvent v2.2 inventories for four different field crops representing Swiss organic and integrated (representing a less intensive conventional production) agricultural practices (Nemecek *et al.*, 2007):

- Wheat grains organic, at farm/CH
- Wheat grains IP, at farm/CH
- Barley grains organic, at farm/CH
- Barley grains IP, at farm/CH
- Soy beans organic, at farm/CH
- Soy beans IP, at farm/CH
- Potatoes organic, at farm/CH
- Potatoes IP, at farm/CH

Fertilizer based nitrogen inputs and fertilizer based N emissions as N_2O, NH_3, and NO_3 within the product related ecoinvent inventories were transformed to amounts of N input and N emissions per ha using the data in Nemecek *et al.* (2005) and Nemecek *et al.* (2007). In organic, slurry and solid manure were assumed to consist of cattle and pig slurry / solid manure respectively and in IP, of cattle slurry / solid manure respectively as described in Nemecek *et al.* (2005). Dilution of slurry was assumed as 1:1.5 (slurry : water) as in the ecoinvent processes. Based on the same fertilizer N inputs and crop yields assumed in the ecoinvent inventories we recalculated N_2O, NH_3, and NO_3 emissions from nitrogen fertilization as described in the following sections and changed these values in the original inventories.

2.1. NH_3 emissions

NH_3 emissions from slurry and solid manure were determined using an emission factor of 20% of applied total ammoniacal nitrogen (TAN). This factor was determined as the upper bound in a recent Swiss study measuring NH_3 emissions from slurry over cropland using improved analytical techniques (Sintermann *et al.*, 2011). TAN values assumed were 0.96 kg/m^3 of slurry and 1.32 kg/t of solid manure in the organic inventories and 0.92 kg/m^3 of slurry and 0.80 kg/t of solid manure in the IP inventories (Nemecek *et al.*, 2005). NH_3 emissions from urea were estimated as 15% and for all other mineral N-fertilizers as 2% from total applied nitrogen (Asman, 1992; ECETOC, 1994). These are the same emission factors as used in the original ecoinvent inventories (Nemecek *et al.*, 2007).

2.2. N_2O emissions

Soil borne N_2O emissions from nitrogen fertilization were estimated using the model by Meier *et al.* (2012), which was developed to better represent the mode of action of organic fertilizers by considering the N-flows in and out of the soil N pool using the IPCC emission factors (IPCC, 2006a). By considering the management induced N turnover processes in the soil the model also integrates nitrogen mineralized from or immobilized in the soil. Deviating from Meier *et al.* (2012) here the amount of management induced nitrogen mineralization from the soil organic N pool was determined by using the SOM-model by Brock *et al.* (2012), in particular by using Equation 1. In combination these models allow for a detailed calculation of the internal N flow based on the N inputs from fertilizers, N fixation, N deposition, and the management induced N turnover in the soil and on the N output by the yield.

$$SON_{MIM} = \frac{N_{PB} - N_{Fix} - N_{Dep} \times NUR_{NDep} - \sum_{i=1}^{n} N_{tot_{Fertilizer\ i}} \times NUR_{NFertilizer\ i}}{NUR_{SONMIM}} + \Delta N_{min} \qquad \text{Eq. 1}$$

SON_{MIM}	management induced nitrogen mineralization from the soil organic N pool [kg N];
N_{PB}	nitrogen in plant biomass [kg N];
N_{Fix}	nitrogen derived from the atmosphere by legumes via symbiotic fixation [kg N] (in case of the four crops analyzed only relevant for soy beans);
N_{Dep}	nitrogen from atmospheric deposition [kg N];
NUR_{NDep}	nitrogen utilization rate [%] of nitrogen from atmospheric deposition;
$NUR_{NFertilizer\ i}$	nitrogen utilization rate [%] of nitrogen from fertilizer i;
NUR_{SONMIM}	nitrogen utilization rate [%] of mineralized nitrogen from the soil;
ΔN_{min}	excessive nitrogen mineralization due to mechanical impact [kg N] (in case of the four crops analyzed only relevant for potatoes).

Equation 1 estimates the management induced N input from the C-N-pool by calculating the N balance between the amount of N built into plant biomass and the N input from fertilizers, atmospheric deposition, symbiotic N fixation, and excessive N mineralization due to intensive mechanical impact as it is the case in potato cultivation. The fact that only a fraction of the N coming from the various pools can be built into plant biomass is considered by pool specific nitrogen utilization rates (Equation 1).

Nitrogen in wheat biomass (straw and grains) was determined using data from a long term field trial comparing organic with conventional production (Gunst *et al.*, 2013). From this trial different N contents for organic wheat and wheat from integrated production were available. Nitrogen contents for the remaining three crops were taken from Flisch *et al.* (2009) not further differentiating between organic and conventional production.

For soy beans N_{Fix} was assumed to be 150 kg N/ha for integrated production and yield adjusted for organic resulting in 144 kg N/ha.

Atmospheric deposition was assumed to be 25 kg N/ha per year for all crops.

Values for the different nitrogen utilization rates and for the excessive nitrogen mineralization in the case of potatoes (50 kg N/ha for organic and IP) were taken from Brock *et al.* (2012). Since in the case of N from symbiotic fixation 100% of the nitrogen is built into plant the nitrogen utilization rate is 1.

Above and below ground residues were estimated on the basis of the formulas given in the IPCC Guidelines on N_2O emissions from managed soils (IPCC, 2006b).

2.3. NO_3 emissions

Emissions of NO_3 may occur from the nitrogen surplus during the cultivation period of a crop plant (NO_3-$N_{short\ term}$, see Equation 3) and after the cultivation period from the plant residues and excessive nitrogen from mineralization due to mechanical impact (NO_3-$N_{long\ term}$, see Equation 4). Total NO_3 emissions from the cultivation of a crop plant are obtained by Equation 2.

$$NO_3 - N_{tot} = NO_3 - N_{short\ term} + NO_3 - N_{long\ term} \quad \text{Eq. 2}$$

$$NO_3 - N_{short\ term} = \left(\sum_{i=1}^{n}\left(N_{tot_{Fertilizer\ i}} \times (1 - NUR_{NFertilizer\ i})\right) + (SON_{MIM} - \Delta N_{min}) \times (1 - NUR_{SONMIM}) + N_{Dep} \times (1 - NUR_{NDep})\right) - (NH_3 - N) - (N_2O_{direct} - N) \quad \text{Eq. 3}$$

$$NO_3 - N_{long\ term} = (N_{PR_{ag}} + N_{PR_{bg}} + \Delta N_{min}) \times Frac_{leach} \quad \text{Eq. 4}$$

NO_3-$N_{short\ term}$	amount of N [kg N] potentially being leached from the production of the crop plant;
NO_3-$N_{short\ term}$	amount of N [kg N] potentially being leached during the cultivation period of the crop plant;
NO_3-$N_{long\ term}$	amount of N [kg N] potentially being leached after the cultivation period of the crop plant by plant;
NH_3-N	ammonia N [kg N] determined as described in section 2.1;
N_2O_{direct}-N	N [kg N] from direct N_2O emissions determined as described in section 2.2.
N_{PRag}	amount of N [kg N] in above ground plant residues
N_{PRbg}	amount of N [kg N] in below ground plant residues
$Frac_{leach}$	fraction of N lost by leaching

We determined the amount of NO_3 leached during the cultivation period of the crop plant from the nitrogen surplus within the respective crop plant system as a proxy for potentially leached N using Equation 3. The nitrogen surplus is determined by the fractions of nitrogen not used by the crop plant from fertilizers, management induced soil-N mineralization (minus the amount of N from excessive mineralization due to mechanical impact (ΔN_{min})), and deposition. If the amount of nitrogen lost as NH_3 and N_2O is subtracted from the surplus, the nitrogen left is basically what can potentially be lost as NO_3. For the calculation of the amount of N potentially leaching from above and below ground plant residues and from N from excessive mineralization due to mechanical impact we used the IPCC (2006b) emission factor of 0.3 ($Frac_{leach}$).

2.4 Impact assessment

The remodeled N emissions per hectare from fertilizer applications were transformed to emission values per kg product again using the crop yields and allocation factors assumed in the ecoinvent inventories. Impacts of global warming, eutrophication and acidification were assessed for the former ecoinvent inventories as well as for the inventories with the remodeled N emissions from fertilizer applications. Impact assessment methods used were IPCC GWP 100a (IPCC, 2007) for global warming and EDIP2003 (Hauschild and Potting, 2003) for eutrophication (N only) and acidification. Relative differences between impacts of crops from organic and integrated production were calculated setting IP as the 100% basis.

3. Results

For all crops, irrespective of organic or IP production, the amount of N supplied by fertilizers was not sufficient to match the required N content in the plant biomass (Table 1). Thus, in most cases a substantial amount of N had to be supplied by the soil C-N-pool besides the inputs from fertilization, symbiotic fixation, and atmospheric deposition. Especially in the case of IP crops the amount of N mineralized from the soil C-N-pool seems fairly high. However, except for the harvested products all N in above and below ground biomass will be returned to the C-N-pool after harvest. The sum of the N inputs from fertilizers, fixation, deposition and soil C-N-pool always exceeded the N content in total plant biomass (Table 1) because – except for N from symbiotic fixation – never a 100% of the N from the inputs is built into plant biomass. Between 7% (soy beans) and 27% (barley) of the N from the inputs remained as surplus depending on the share of the different N input sources and their nitrogen utilization rates (see Eq. 1). Due to yield differences between the crops from organic production and IP the amount of N in plant biomass was 1.6 to 1.7 times lower in organic crops expect for soy beans where the N content in plant biomass was only 4% lower in organic soy bean which corresponded well with the yield gap of also 4%. For wheat, barley, and potatoes the amount of N mineralized from the soil C-N-pool to meet the plants N demand was approximately 4 to 12 times higher in IP production compared to organic. In contrast for the same crops amounts of N supplied by fertilizers were only 1.2 to 1.5 times higher in IP compared to organic (Table 1).

Table 1. N content in plant biomass vs. N inputs of the four crops analyzed.

	Wheat organic	**Wheat IP**	**Barley organic**	**Barley IP**	**Soy beans organic**	**Soy beans IP**	**Potatoes organic**	**Potatoes IP**
N content in plant biomass[1] [kg N/ha]	136	230	99	159	220	230	89	137
N input from fertilizers [kg N_{tot}/ha]	122	143	98	121	20	27	84	127
N input from fixation [kg N/ha]	0	0	0	0	144	150	0	0
N input from deposition [kg N/ha]	25	25	25	25	25	25	25	25
N mineralized from soil C-N-pool [kg N/ha] (without ΔN_{min})	35	134	13	69	49	48	2	23
N surplus [kg N/ha][2]	46	72	37	56	18	20	22	38
NH_3-N remodeled [kg/ha]	11.0	6.4	8.9	5.8	1.2	2.1	4.1	6.4
N_2O-N_{direct} remodeled [kg/ha]	1.4	2.2	1.1	1.7	0.7	0.7	1.0	1.5
N_2O-$N_{indirect}$ remodeled [kg/ha]	0.4	0.7	0.3	0.5	0.2	0.3	0.3	0.5
NO_3-$N_{short\ term}$ remodeled [kg/ha]	33.5	63.4	27.2	48.3	15.5	17.4	16.7	29.8
NO_3-$N_{long\ term}$ remodeled [kg/ha]	9.5	18.2	6.5	11.6	13.2	13.6	21.0	22.2
Delta N [kg/ha][3]	1	-0.1	-0.2	-0.1	0	0	1	-0.1

[1] Total plant biomass including harvested products and above and below ground plant residues.
[2] N in plant biomass minus fertilizer N, minus N from fixation, minus N from deposition, minus N mineralized.
[3] N surplus minus NH_3-N, minus N_2O-N_{direct}, minus NO_3-$N_{short\ term}$.

The remodeled N emissions from fertilizer applications on a per ha basis showed the least differences for ‘Wheat IP’ (Table 2). In the case of ‘Barley IP’ NH_3 and N_2O emissions also differed only slightly whereas NO_3 emission were 1.6 times lower after remodeling. For ‘Potatoes IP’ NH_3 emissions were 2.3 times lower for the remodeled emissions whereas N_2O and NO_3 emissions changed only little after remodeling. Changes in the inventories of organic crops were generally higher than for the respective IP crops (Table 2).

Table 2. Fertilizer based N emissions per ha from the ecoinvent inventories and the remodeled values for the four crops analyzed.

	Wheat organic	**Wheat IP**	**Barley organic**	**Barley IP**	**Soy beans organic**	**Soy beans IP**	**Potatoes organic**	**Potatoes IP**
N emissions in ecoinvent inventories								
NH_3 [kg/ha]	33.93	9.06	29.22	9.63	3.45	5.70	15.71	18.31
N_2O [kg/ha]	4.93	6.07	4.69	4.29	6.96	7.34	3.40	4.07
NO_3 [kg/ha]	400.27	331.57	406.00	425.72	194.34	200.26	292.21	261.85
remodeled N emissions								
NH_3 [kg/ha]	13.41	7.79	10.78	6.99	1.41	2.55	5.03	7.80
N_2O [kg/ha]	2.93	4.47	2.24	3.41	1.40	1.55	2.10	3.12
NO_3 [kg/ha]	190.15	361.37	149.44	265.00	127.23	137.45	167.03	230.54

Comparing the crops from organic with the crops from IP production assessing the original ecoinvent inventories organic wheat and organic soy beans showed lower impacts for global warming (-9%, -12% respectively) than the respective crops from IP production (Figure 1). Impacts for global warming from organic barley and organic potatoes were higher (+17%, +16% respectively) than from IP crops. Impacts from eutrophication were higher for all organic crops (wheat: +108%; barley: +72%, potatoes: +39%) except for soy beans showing 26% lower impacts for eutrophication than IP soy beans. Impacts for acidification were manifold higher in organic cereals than in the respective cereals from IP production (wheat: +271%; barley: +241%). Also organic potatoes had higher impacts for acidification (+32%), whereas organic soy beans showed also lower impacts for acidification (-38%) than IP soy beans (Figure 1).

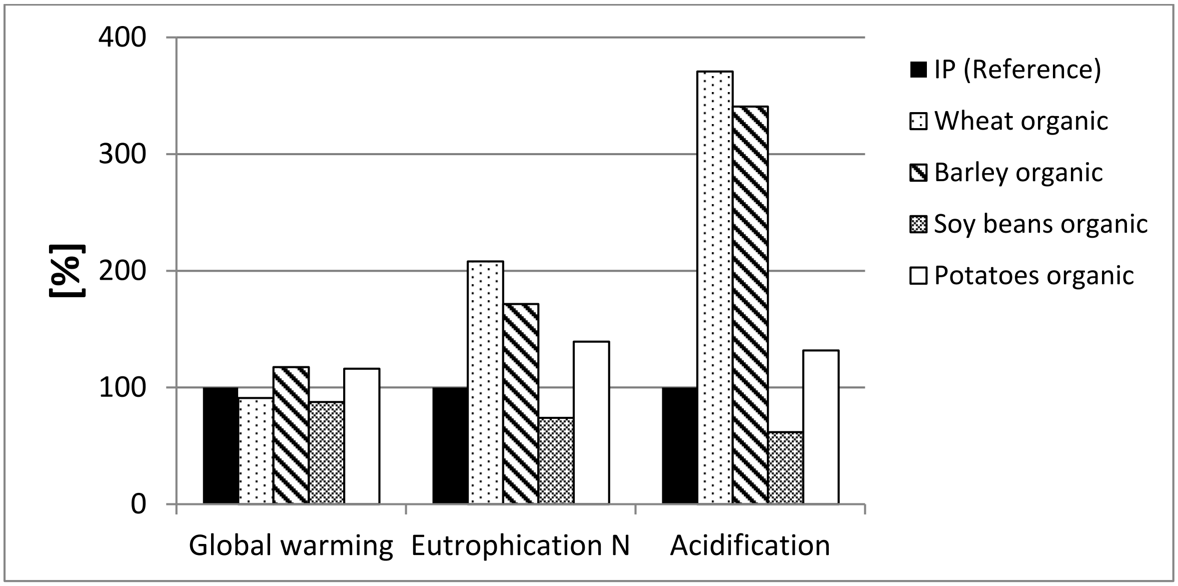

Figure 1. Relative differences between organic crops and the respective crops from IP production for global warming, eutrophication, and acidification per kg of grains, pulses and tubers respectively assessing the original ecoinvent inventories.

Comparing the four organic crops with their counterparts from IP production assessing the inventories with the remodeled N emissions from fertilizer applications all organic crops but potatoes showed lower impacts for global warming (wheat: -24%; barley: -17%; soy beans: -23%) (Figure 2). Impacts for global warming of organic potatoes were 8% higher than for the potatoes from IP production. Again impacts for eutrophication were lower (-40%) for organic soy beans. However, after remodeling N emissions, impacts for eutrophication of the other three organic crops were comparable to the impacts of the respective IP crops (wheat: -4%; barley: +6%; potatoes: +1%). Impacts for acidification of the organic cereals were still clearly higher than for the IP cereals (wheat: +89%; barley: +81%; Figure 2) but the difference became much smaller compared to the original ecoinvent inventories (Figure 1). Also the difference in the impacts for acidification of organic potatoes compared to IP potatoes became smaller in the remodeled inventory (+13%). Finally, impacts for acidification of organic soy beans were also lower (-40%) compared to IP soy beans in the inventory with the remodeled N emissions (Figure 2).

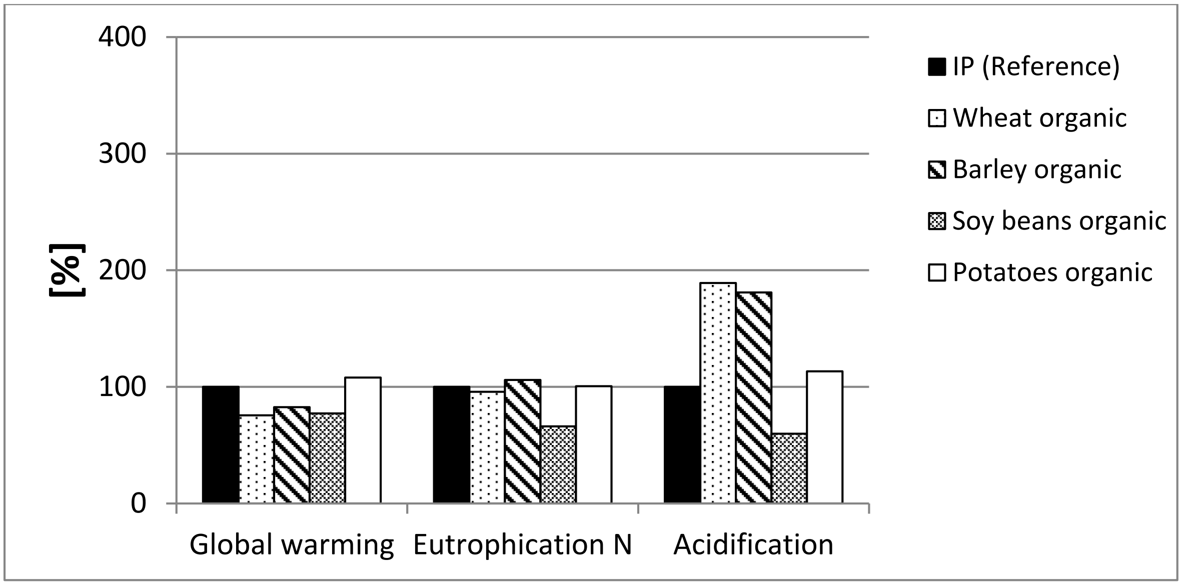

Figure 2. Relative differences between organic crops and the respective crops from IP production for global warming, eutrophication, and acidification per kg of grains, pulses and tubers respectively assessing the inventories with the remodeled N emissions from fertilizer applications.

4. Discussion

4.1. N emission modeling

The remodeled NH_3 emissions from fertilizer applications on a per ha basis were lower for all crops than the emissions in the original ecoinvent inventories. This can be explained by the different ammonia emission rates from slurry that were used in the estimations of the remodeled values and which are based on recent ammonia emission measurements from slurry application in Switzerland (Sintermann *et al.*, 2011). However, reductions in NH_3 emissions were always higher in organic crops after remodeling. This is because in organic crop production nitrogen is solely applied as organic fertilizers (slurry and solid manure for the crops analyzed in this study) having higher NH_3 emissions than mineral fertilizers whereas the crops in integrated production are mainly fertilized by mineral fertilizers.

After remodeling, N_2O emissions became less on a per ha basis for all organic crops except for potatoes. For the IP crops N_2O emissions increased after remodeling except for soy beans where the same decrease could be observed as for organic soy beans. For soy beans similar amounts of nitrogen fertilizers were applied, which explains the similar amounts of N_2O emissions. Further, in soy beans the greatest difference between N_2O emission values in the original ecoinvent inventory and the remodeled values could be observed. This can be explained by the modeling of the N_2O emissions according to IPCC guidelines 1996 in the original ecoinvent inventory (Nemecek *et al.*, 2007) where the amount of N from fixation was added to the amount of N from fertilizers. The higher N_2O emissions for organic potatoes after remodeling is due to the additional N from excessive nitrogen mineralization due to mechanical impact, which the model used here accounts for. The higher N_2O emissions after remodeling in wheat, barley, and potatoes from integrated production can be explained by inclusion of the management induced N mineralization from the soil C-N-pool by which the full N turnover in the modeling of N_2O emissions is considered. Relative to the N input from fertilizers the amount of N from the soil C-N-pool was larger in wheat, barley, and potatoes from integrated production than in the respective organic crops. This also explains why the differences in N_2O emissions between these organic and the respective IP crops became larger after remodeling.

Since NO_3 emissions are difficult to model reliably we estimated the total amount of leachable N on the basis of the nitrogen balance which resulted in lower NO_3 emissions per ha for all crops except potatoes irrespective of organic or IP. This indicates that in the original ecoinvent inventories NO_3 emissions are overestimated. Especially for organic fertilizers the NO_3 model used within the original ecoinvent inventories seems to produce values that are too high. In contrast the higher NO_3 emissions in potatoes after remodeling can again be explained by the additional N from excessive nitrogen mineralization due to mechanical impact which was considered for the remodeled values.

4.2 Impact assessment

In absolute terms impacts for global warming, eutrophication, and acidification became lower after remodeling for all crops irrespective of organic or IP with the exception of impacts for eutrophication in IP wheat resulting in higher absolute impacts after remodeling. The lower relative impacts for global warming per kg of organic wheat, barley and soy beans compared to the respective IP crops after assessing the inventories with the remodeled N emissions from fertilization can be explained by the considerably lower N_2O emissions. In the case of potatoes the relative difference in impacts for global warming between organic and IP potatoes became smaller after remodeling with still slightly higher impacts for global warming in organic.

Assessing the impacts for eutrophication on the amount of leachable N estimated on the basis of the N balance shows that impacts for eutrophication on a per kg of product basis of organic wheat, barley, and potatoes is comparable to the impacts for eutrophication of the corresponding crops from integrated production. Impacts for eutrophication of 1 kg of soy beans were already lower in the original ecoinvent inventories. However, the relative difference between organic and IP soy beans increased after remodeling because the relative difference of NO_3 emissions on the per ha basis increased.

Impacts for acidification on a per kg of product basis of organic crops decreased substantially when assessing the inventories with the remodeled N emissions. However, especially for organic wheat and barley impacts for acidification were still considerably higher than for the respective crops from integrated production. This is due to the sole use of organic fertilizers in organic crop production. Even after remodeling NH_3 emissions from organic wheat and barley were higher on a per ha basis than from the IP crops. Though, with improved slurry application techniques and management practices NH_3 emissions from organic fertilizers could be further reduced resulting in a further decrease in impacts for acidification from organic crops.

5. Conclusion

This analysis shows that N emissions from fertilization in LCA inventories of organic and conventional agricultural products are not well adapted to the respective farming system. In particular the crop specific changes in the soil C-N-pool need to be considered in the emission modeling to better represent N_2O emission from organic fertilizers and to take account of the total nitrogen turnover from crop production.

Calculating N emissions based on the actual N flow in the respective farming system leads to a more precise differentiation of impacts for global warming, eutrophication, and acidification between organic and conventional products. By that higher accuracy in N modeling from fertilization helps to improve the differentiation of environmental impacts of products from different agricultural systems such as organic and conventional agriculture and allows for conclusions on their N efficiency.

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A statistical approach to deal with uncertainty due to the choice of allocation methods in LCA

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Unresolved debates on the application of allocation methods constitute a major source of uncertainty in Life Cycle Assessment (LCA) results. Some ways to deal with this issue are standardization, sensitivity analysis and scenario modeling. Standardization reduces uncertainty, while the latter two methods serve to analyze the system using one allocation at a time in order to show a range of possible outcomes. However, the full range of outcomes given all possible choices has never been shown. This can be misleading when evaluating the environmental impacts of a production system, particularly when comparing two systems. We investigate the use of Monte-Carlo simulations as a statistical approach for analyzing uncertainty propagation to the LCA outcomes, depending on the choice of allocation method. In this approach, a probability of occurrence is assigned to each allocation method, for each multi-functional process in the system. The allocation methods are sampled according to the probability of occurrence using Monte-Carlo simulations. The range and frequency of LCI outcomes was analyzed with and without this approach, including data uncertainty in both cases. The LCI results show the influence of many possible allocation choices and combinations in a system besides the influence of data uncertainty. Further implementation for more complex systems and broader LCI and LCIA outcomes is required.

Keywords: uncertainty in LCA, allocation, data, Monte-Carlo

1. Introduction

Methodological choices in all phases of Life Cycle Assessment (LCA) are unavoidable and are a potential source of uncertainty. A typical example of such choices is the choice between different methods to solve multi-functionality of unit processes. Generally, practitioners choose ambiguously between the following three types of methods to solve this issue: substitution (avoided burden), system expansion or partitioning, and within partitioning between different principles, e.g., mass, energy, or economic value[1]. This ambiguous choice could be a major source of uncertainty in LCA results since, as mentioned by Guinée et al. (2004): *"... it is impossible to have an ultimate best solution accepted by everybody for a problem that is an artefact of wishing to isolate one function out of many"*. In fact, several studies have shown the influence of the choice of allocation method in the dispersion of LCA results and e.g. policy implications (van der Voet et al. 2010; Wardenaar et al. 2012), and this is one of the reasons why ways to deal with uncertainty due to the choice of allocation method have been a key topic of discussion and development in the LCA arena in recent years.

Until today, the main approaches for accounting for uncertainty introduced to LCA results by the choice of allocation method include: standardization, critical reviews (peer review), sensitivity analysis, and scenario modelling (Björklund 2002). Standardization refers to consensus seeking on which allocation methods to use, and which procedures to follow if multi-functionality is encountered in LCA. Uncertainty due to allocation choices is reduced with standardization, but it does not explicitly lead to LCA results that account for uncertainty due to the choice made. In 1998, ISO (International Standardization Organization) provided guidelines to solve multi-functionality[2] (ISO 2006), which have become more widely applied by practitioners nowadays.

Another method to account for uncertainty due to the choice of allocation methods is sensitivity analysis. This method studies the influence that input parameters (independent) have on output parameters (dependent) (Björklund 2002). A sensitivity analysis that shows the influence of the choice of allocation method (independent parameter) on the LCA outcomes e.g. as Life Cycle Inventory (LCI) or as Life Cycle Impact Assessment (LCIA), is usually implemented by means of scenarios using different allocation methods. This type of sensitivity analysis approach is applied by most LCA practitioners and is referred to as scenario analysis (Björklund

[1] From this point on substitution, system expansion and partitioning (including different principles) are referred to as allocation methods

[2] ISO guidelines for allocation procedure are as follows: first, allocation should be avoided by substitution or system expansion where possible; second, where avoiding allocation is not possible physical partitioning should be applied and where this is not possible economic allocation should be applied. ISO also requires a sensitivity analysis where more than one allocation method seems applicable.

2002). Scenario analysis is still ambiguously defined in LCA because the scenarios design, i.e. which allocation methods to choose and for which multi-functional processes, remains vague and leaves room for debate.

Beside uncertainty introduced by methodological choices, other sources of uncertainty exist in LCA and perhaps the most recognized and addressed one is data uncertainty (Björklund 2002). Data uncertainty at the unit process level has three sources of uncertainty according to Henriksson et al. (2013): representativeness, inherent uncertainty and spread due to averaging of data. The authors developed a practical protocol quantifying the total dispersion of unit process data due to these three uncertainty sources. The explicit description of the input parameters, using distributions, mean values and standard deviations, is systematically facilitated by the protocol of Henriksson et al. (2013), which goes beyond the post-normal pedigree matrix based treatment of data uncertainty in LCA (Weidema & Wesnæs 1996; Funtowicz & Ravetz 1990).

Subsequently, a method has to be selected to propagate these data uncertainties into output uncertainties through the LCA model. For this, several methods exist and have been used in LCA (Heijungs & Lenzen 2014). Among the most popular ones are statistical methods, which include sampling methods such as Monte Carlo (MC) simulations. MC is becoming more and more commonly used in LCA, as it relies on computing capacity, which has increased in time (Heijungs & Huijbregts 2004). Other methods have also been studied, such as analytical uncertainty propagation using e.g. Taylor series expansions. Taylor expansions and analytical methods have been found to lead to similar results as MC simulations, while providing the contribution to uncertainty from each parameter and strongly reducing calculation time (Hong et al. (2010), Heijungs et al. (2005)). Nonetheless, more recently (Heijungs & Lenzen 2014) showed that both analytical and sampling methods are equally important and required for a good analysis, particularly sampling methods when uncertainties are large.

Analytical propagation of uncertainty for allocation factors was developed by Jung et al. (2013). The authors developed a method integrating allocation factors in matrix based LCA calculations and propagating the uncertainty in allocation factors to the LCA outcomes. In other words, the choice for an allocation method is made, but the factors themselves are described as uncertain input parameters leading to uncertainty in the LCA outputs. This method described by Jung et al. (2013) to address uncertainty in allocation factors and in data together is the most recent attempt in literature to deal with both sources of uncertainty together and uses an analytical method for propagation of uncertainty to LCI and LCIA results, i.e. a first-order approximation.

Despite these efforts to deal with data related and allocation related uncertainties, no statistical sampling method exists yet, to our knowledge, to simultaneously propagate through LCA, uncertainties due to data and to the choice of allocation methods. The aim of this study was to develop such a method, so that it can be used in LCA calculations and software. This study uses the CMLCA software as a testing software and implements an illustrative simple case study. Finally, we will simultaneously explore spread of LCI results when propagating uncertainty due to the choice of allocation method and due to unit process data uncertainties using MC simulations.

2. Methods

2.1. Uncertainty sources and propagation methods

Focusing on unit process and allocation uncertainties, we can distinguish the following sources of uncertainty: i. uncertainty in data at the unit process level (covering representativeness, inherent uncertainty and spread); ii. uncertain allocation factors per se (e.g. fluctuating prices lead to uncertain allocation factors for economic value allocation) and iii. choice of allocation methods (substitution, system expansion or partitioning). The main methods for propagating these uncertainty sources into LCA outcomes are: a) scenario analysis, b) analytical methods and c) sampling methods.

Table 1 shows the field of options that results from combining the previously described uncertainty sources and propagation methods. The table also shows some of the studies that have addressed particular sources with particular methods and combinations of both. Table 1 indicates that there are many studies that deal with allocation choices by applying scenario analysis, and that there are no studies that propagate uncertainty due to the choice of allocation method in an analytical way (perhaps because of its impossibility) and finally, that this study is the first one propagating uncertainties due to data and to the choice of allocation method in a simultaneous way.

Table 1. Field of options and examples of literature studies combining different propagation or analysis methods for addressing selected sources of uncertainty in LCA

Propagation / analysis method — Uncertainty source	a) Sensitivity Analysis / scenario Analysis	b) Analytical Methods	c) Sampling Methods
i. Unit process data	e.g. Middelaar et al. (2012) van der Harst & Potting (2014)	e.g. Heijungs et al. (2005) Hong et al. (2010) Imbeault-Tétreault et al. (2013) Jung et al. (2013)	e.g. Guo & Murphy, (2012) Gregory et al. (2013) Imbeault-Tétreault et al., (2013) Sonnemann et al. (2003) ***This Study (Option 2 and 3)***
ii. Allocation factors	e.g. Ardente & Cellura (2012) Huppes (1993)	e.g. Jung et al., (2013)	-
iii. Choice of allocation method + principles	e.g. Ardente & Cellura (2012) Ayer, N.W. et al. (2007) González-García et al. (2012) Jeroen B Guinée & Heijungs (2007) Heijungs & Jeroen B Guinée (2007) van der Harst & Potting (2014) Luo et al. (2009) Svanes et al. (2011) Wardenaar et al. (2012) ***This study (Option 1 and 2)***	-	***This study (Option 3)***

The options we analyze in this study are the following:

Option 1: Scenario analysis for different allocation methods without statistical propagation for data uncertainty

This option consists of the classic scenario analysis to evaluate the influence of the allocation method on the LCA results. In this case, no data uncertainty is considered and only different allocation methods are applied to some multi-functional processes in the system, resulting in scenarios with punctual values for LCI outcomes given different allocation methods chosen for the calculations. For a large system with several multi-functional processes, it is unrealistic to analyze all the possible scenarios. The total number of scenarios is equal to a^n, where *n* is the number of multi-functional processes in the system and *a* is the number of allocation methods possible per process. Usually, LCA practitioners undertake a contribution analysis to help them decide which processes to analyze further and then they only create scenarios for the most important multifunctional processes, thereby significantly reducing the amount of scenarios. However, this approach can be misleading as the multi-functional process itself may have no direct emissions, but still have a big influence on the LCA results via upstream inputs.

Option 2: Scenario analysis for fixed allocation settings with statistical propagation for data uncertainty

Data at the process level is uncertain. In this option, we will apply the protocol of Henriksson et al. (2013) to quantify unit process data dispersions due to inherent uncertainty, unrepresentativeness and spread, and derive data distributions to be used as input parameters for the LCA model. Furthermore, we use MC simulations as a propagating method of the unit process data uncertainties and calculate LCI results based on the MC simulations outputs. For the choice of allocation method, fixed choices are specified, as is done in the scenario analysis.

Option 3: Statistical propagation for choice of allocation method and data uncertainty

This option propagates both data and choice of allocation uncertainties simultaneously, using MC simulations based on data distributions as in option 2 and probabilities of occurrence for the choice of allocation method as will be described in section 2.2. This option represents the novel contribution of this study. We will illustrate the possible advantages of this new method by showing the outcomes of the options described above for a simplified agricultural system i.e. rapeseed oil production in Northern Europe.

2.2. Implementation of a statistical propagation method for uncertainty due to the choice of allocation method in this study

For a multi-functional unit process, several *allocation methods* can be applied in order to solve multi-functionality. For partitioning-type of allocation methods, *allocation factors* are defined as the fraction that divides the non-functional economic and environmental flows to the functional flows (i.e. the co-products) of a multi-functional process (Guinée et al. 2004). For each *multi-functional unit process* in the system, allocation factors are defined to be able to solve the matrix algebra (Heijungs & Suh, 2002) behind LCA modeling. Typically, the sum of all allocation factors for each allocation method is equal to 1. This is, in very general terms, the normal parameter definition for partitioning type of methods using different principles such as mass, energy content and economic allocation. For substitution type of allocation, equivalent flows are found for the co-product to be substituted by another product(s), as well as the ratio of substitution.

In this study we introduce a parameter named the *probability of occurrence* (p) (as a percentage) for each allocation method which ranges from 0 to 100% and which sum for all p-values equals 100%. This means that for each multi-functional process, possible allocation methods are: method 1, 2, 3, 4 etc., which have an associated p_1, p_2, p_3, p_4 etc. percentage of occurrence between 0 and 100% and all together add up to 100%. Together they describe *ranges of occurrence* for each method i.e. from 0 to p_1 for method 1, from p_1 to $p_1 + p_2$ for method 2, from $p_1 + p_2$ to $p_1 + p_2 + p_3$ for method 3, until from $p_1 + p_2 + p_3 + p_a$ to 1 for method a.

In principle, to propagate the uncertainty due to the choice of allocation method through the LCA model into the LCA outcomes, a random real number R is taken from a continuous uniform distribution U (0,100), each time a multi-functional unit process is encountered in the calculation of the system, for every Monte Carlo simulation. This value (R) is evaluated for the ranges of occurrence, and this is how the allocation method and factors are defined for each individual simulation. If for example four allocation methods are defined for a specific multi-functional process*:*

$$
\begin{array}{cl}
 & R \sim U(0{,}100) \\
0 \le R \le p_1 & \Rightarrow \text{ use allocation method 1} \\
p_1 < R \le p_1 + p_2 & \Rightarrow \text{ use allocation method 2} \\
p_1 + p_2 < R \le p_1 + p_2 + p_3 & \Rightarrow \text{ use allocation method 3} \\
p_1 + p_2 + p_3 < R \le 100 & \Rightarrow \text{ use allocation method 4}
\end{array}
$$

By repeating the evaluation of R for a large number of runs, for example using MC simulations, several choices for an allocation method for each multi-functional process are taken into account, leading to the propagation of uncertainty due to the choice of allocation method to the LCA results. In each single MC simulation the allocation method for each unit process is selected using the ranges of occurrence for each method, and the results are calculated using such choice. The higher the sample the more combinations of allocation choices are taken into account in the results.

When analyzing a system with a relative large number of multi-functional processes the previously described propagation method becomes very powerful, as a large amount of scenarios can be reproduced minimizing the amount of inputs. For example, if the number of multi-functional processes is 10 and two allocation methods are possible for each process, then 1024 combinations exist if all allocation methods are combined for all multi-functional processes. A complete scenario analysis should show the outcomes for all the combinations. However, that is rarely the case in LCAs as it is a very time consuming process. With the proposed method 1024 combinations can be covered while defining the ranges of occurrence for each method per unit process only one time.

An example of the input parameters necessary for the method to propagate uncertainty due to the choice of allocation method as was described are shown in Table 2. For one multi-functional process four allocation methods are displayed in Table 2. These are the methods currently implemented in CMLCA. For each method the corresponding allocation factors for each co-product are shown. The sum of these factors for one method is equal to 1. For example, partitioning-2, has allocation factors 0.7 and 0.3 for co-product one and two of this example multi-functional unit process. Furthermore, the percentage of occurrence for each method is also defined. For the example, substitution occurs 2.5%, surplus 2.5%, partitioning-1 25% and partitioning-2 70%. Together the percentage of occurrence for the four allocation methods adds up to 100% as well. Thus, the ranges of occurrence for each allocation method in the example of Table 2 are: from 0 to 2.5% substitution, from 2.5% to 5% surplus, from 5% to 30% partitioning-1 and from 30% to 100% partitioning-2.

Table 2. Parameters for different allocation methods for a multi-functional unit process. This table displays the allocation methods implemented in the CMLCA software only

	One Multi-functional Unit Process			
Allocation Method	**Co-product**	**Allocation Example**	**% of occurrence (*p*)**	**Example**
1.Substitution	Co-product 1	1 kg rapeseed cake replaced by 1.5 kg peas	P_1	2.5%
	Co-product 2	-		
2.Surplus	Co-product 1	0	P_2	2.5%
	Co-product 2	1		
3.Partitioning-1	Co-product 1	0.55	P_3	25%
	Co-product 2	0.45		
4.Partitioning-2	Co-product 1	0.7	P_4	70%
	Co-product 2	0.3		
a. Allocation method	Co-product 1	C_1	P_a	P_a %
	Co-product 2	C_2		
	Co-product Y	$1-C_1-C_2$		

2.3. Illustrative case study

We have implemented in CMLCA version beta 5.2 a simple system: Rapeseed Oil production in Northern Europe, similar to Wardenaar et al., (2012). We focus on three key processes (1) cultivation, (2) transport to mill and (3) rapeseed oil extractions by cold pressing of rapeseed. Process one produces straw and rapeseed and process three rapeseed oil and rapeseed cake. Thus both processes are multi-functional and require allocation between the co-products. The system is described in Figure 1. The functional unit is 1 kg of rapeseed oil at mill and the system includes the production, storage and transport of the main inputs to these three key processes. For the description of the background processes, ecoinvent data from version 2.2 is used (http://www.ecoinvent.org/database/). Also, background processes are assumed to be allocated as done by the ecoinvent database and this assumption remain constant for all the options analyzed in this study.

For the two multi-functional processes, we assume two allocation methods are applicable i.e. surplus and mass for rapeseed cultivation and energy content and economic values for oil extraction. The allocation parameters defined for the case study are shown in Table 3. For *option 1* we use the four combinations possible using two multi-functional processes with two allocation methods each. This leads to four fixed allocation scenarios (Table 3) and the LCI results of this option are presented as point values given that no data uncertainty is assumed. For *option 2* we use the same fixed allocation scenarios as option 1, but in this case data uncertainties are propagated to LCI outcomes with MC simulations. We used a sample size of 700 simulations for the four fixed allocation scenarios. Table 3 shows that the percentage of occurrence assigned to each method is equal to 100, simply because choosing for one method corresponds to 100% probability of occurrence of that method, according to the method described in section 2.2. This holds for both options 1 and 2. Finally, for *option 3* we use 50% probability of occurrence of the allocation methods in both multi-functional processes, because this allows an equal chance of occurrence for all the methods. We ran option 3 for 2000 MC simulations which is different than the number of simulations in option 2 a choice made for practical reasons.

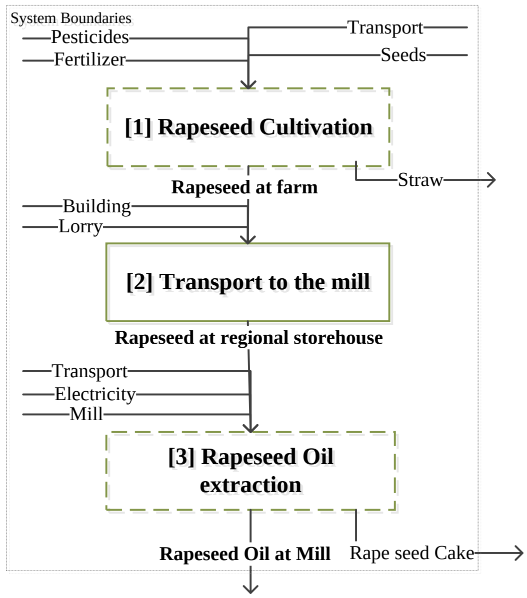

Figure 1. System for rapeseed oil production in Northern Europe. Boxes represent processes, dashed boxes are multi-functional.

Table 3. Allocation factors and probability of occurrence of each allocation methods for the two multi-functional processes in the rapeseed oil production system

	Multi-functional Process		Probability of occurrence p (%) for the different options				
	[1] Rapeseed cultivation		Option 1(no data uncertainty) and 2 (with data uncertainty, 700 MC)				Option 3 (data + choice of allocation method uncertainty)
Allocation Method	**Co-product**	**Allocation factor**	**Fixed Allocation1**	**Fixed Allocation2**	**Fixed Allocation3**	**Fixed Allocation4**	**2000 MC**
Surplus	Straw	0	100	0	0	100	50
	Rapeseed	1					
Mass	Straw	0.43	0	100	100	0	50
	Rapeseed	0.57					
	[3] Rapeseed oil extraction						
Energy Content	Rapeseed Oil	0.55	100	100	0	0	50
	Rapeseed Cake	0.45					
Economic value	Rapeseed Oil	0.7	0	0	100	100	50
	Rapeseed Cake	0.3					

3. Results

Results for option 1 are given in Table 4 and for option 2 and 3 in Figure 2. Option 1 shows the LCA results for a scenario analysis for different allocation methods without statistical propagation for data uncertainty. The results presented in Table 4 only show carbon dioxide emissions to air per kg of rapeseed oil at mill for the four fixed allocation scenarios. Emissions range from 0.7 to 1.19 kg CO_2 / kg of rapeseed oil depending on the allocation settings chosen for the two multi-functional processes under study. As expected, for those allocation scenarios in which rapeseed and rapeseed oil get a higher allocation factor (fixed allocation 1 and 4), the emissions per kg of rapeseed oil are higher.

Table 4. Carbon dioxide emissions to air per kg of rapeseed oil for fixed allocation (Table 3) and no data uncertainty propagation (Option 1)

Allocation Settings	CO_2 (kg CO_2 / kg of rapeseed)
FixedAllocation1	0,94
FixedAllocation2	0,70
FixedAllocation3	0,89
FixedAllocation4	1,19

Figure 2 shows a series of histograms that corresponds to the results for carbon dioxide emissions to air for option 2 and 3. The outcomes for option 2 correspond to the histograms for the four possible fixed allocation combinations with statistical propagation for data uncertainty (i.e. dashed lines in Figure 2). The outcomes for option 3 correspond to the histogram including statistical propagation of the choice of allocation method together with statistical propagation of data uncertainty (i.e. continuous line in Figure 2).

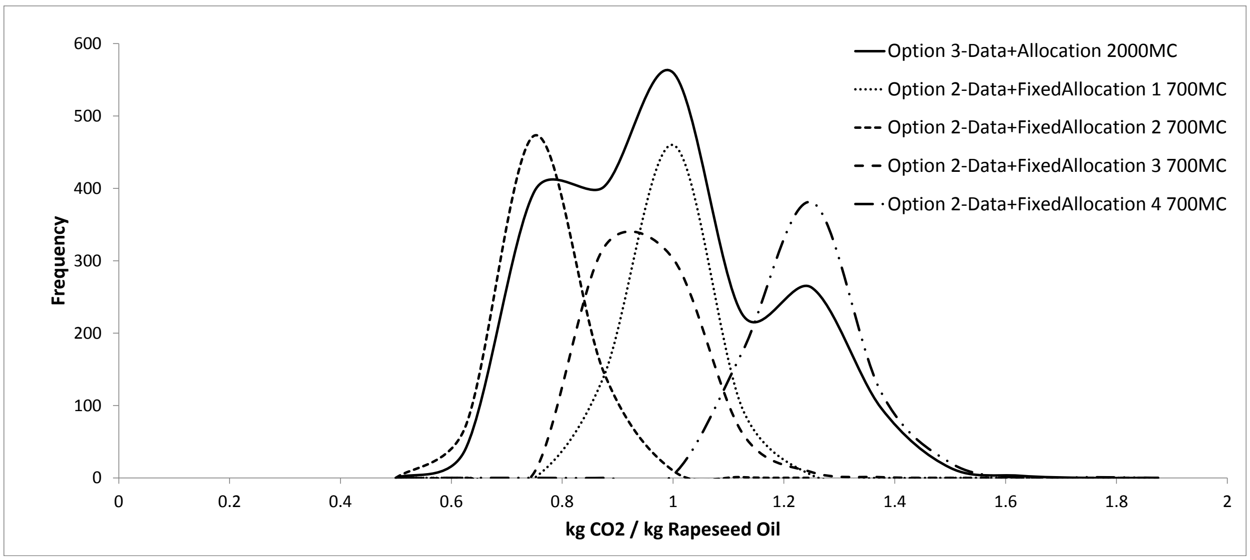

Figure 2. LCI results for carbon dioxide emissions to air per kg of rapeseed oil under four fixed allocation settings (Table 3) and for random allocation choice using MC simulations to propagate uncertainty

4. Discussion

The method presented here to simultaneously propagate uncertainties in data and due to the choice of allocation methods through an LCA, is based on the introduction of the probability of occurrence of each allocation method for a multi-functional process. This parameter can be used by assigning equal probability of occurrence to all allocation methods for a multi-functional process, as shown in the illustrative case of this study. Assuming this, helps to propagate, with equal chances, the uncertainty around the fact that one can choose for different allocation methods in different multi-functional processes. Therefore, assigning equal probability of occurrence to the allocation method per process, is expected to have an influence on the size of the peaks of the histograms i.e. the frequencies with which an outcome is observed, but not on the ranges of the outcomes. The shape of the histograms also depends on the sample size of the MC simulations which at the end determines the distributions of the LCI results.

Also, the use of this parameter allows a full statistical propagation of uncertainty with a large sample size to make sure all methods are sampled more or less equally. It can be argued, nevertheless, that this method could be intensive in terms of computing capacity requirements as it uses MC simulations as a propagating method. Analytical methods do not yet exist - despite recent efforts in the field -, which could be a less computing intensive alternative to MC simulations as proposed in this study, for propagation of the choice of allocation methods if possible at all.

The results showed how scenario analysis for allocating different multi-functional processes in order to explore the dispersion of the LCI outcomes i.e. option 2, leads to a similar range of outcomes for carbon dioxide emissions to air than option 3 which takes into account data uncertainty and allocation choice simultaneously in MC simulations. This is an indication of the strength of the method presented here that allows capturing in LCI results the influence of all possible allocation choices in a system next to the influence of data uncertainty without actually making an explicit choice for one or another method per process. Also, showing the likelihood for which an outcome can be expected given many possible combinations of input parameters and allocation methods for a specific system is another strength of the proposed method. Besides, both option 2 and 3, lead to a broader range of CO_2 emissions than when no data uncertainty is taken into account in the LCA, showing the importance of accounting for data uncertainty as emphatically pointed out by many other studies.

Further research is required to evaluate the method shown here for systems with a large number of multi-functional processes and broader LCI outcomes too. Finally, in the future, the method could be expanded to deal with uncertainties due to other sources not yet addressed in this study.

5. Conclusion

Unresolved debates on the application of allocation methods constitute a major source of uncertainty in LCA results. The full range of outcomes given all possible choices for allocation methods and combinations in a system with several multi-functional unit processes is hardly shown.

We propose the use of Monte-Carlo simulations as a statistical approach to simultaneously propagate uncertainties due to data and to the choice of allocation methods to LCA outcomes. For this purpose the probability of occurrence was introduced and assigned to each allocation method, for each multi-functional process in a system.

The distribution of LCI outcomes was analyzed with and without the previous approach and in both cases including data uncertainty and we conclude that the proposed method enables, in a relatively simple way i.e. with few additional parameters definition and modest computational calculation capacity, to propagate uncertainties due to the choice in allocation method and data uncertainty to the LCA results while not requiring an actual choice for one or another allocation method. Further implementation for more complex systems is required.

6. Acknowledgements

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Goal and Scope Definition for Life Cycle Assessment of Integrated Multi-Trophic Marine Aquaculture Systems

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ABSTRACT

Integrated Multi-Trophic Aquaculture (IMTA) has been regarded as an environmental management concept that can minimize the environmental impacts of conventional monoculture aquaculture farms while expanding their economic base. These characteristics of IMTA systems hold as long as (1) there are no significant trade-offs between the discharge reduction and the life cycle environmental impacts of the IMTA infrastructure; and (2) the co-cultured species in the IMTA system constitute commercial species produced at scales that yield a profit. In this study we use Life Cycle Assessment (LCA) to analyze the environmental trade-offs between monoculture and IMTA systems. We do this by posing two questions: 1) what are the trade-offs for small and medium enterprises (SMEs) considering to move from monoculture aquaculture practice towards IMTA; and 2) what are the trade-offs comparing IMTA species with their conventional monoculture alternatives. Our hypothesis states that the level of integration of the productive activities of the co-cultured species (e.g., bivalves, echinoderms and algae) into the existing monoculture (e.g., finfish) productive activities will determine the magnitude of the trade-offs. Quantification of trade-offs is ongoing as part of the EU FP7 IDREEM (Increasing Industrial Resource Efficiency in European Mariculture) project. We here report our first conceptual results consisting of a scoping framework to be further applied in the project.

Keywords: Sustainable fish production, aquaculture, IMTA, LCA

1. Introduction

Aquaculture, which includes marine fish farming and in this paper refers exclusively to marine systems, faces increasing pressures as demand for seafood products grows while traditional wild fisheries are in decline (FAO, 2012). Integrated Multi-Trophic Aquaculture (IMTA) is the combined cultivation of multiple commercially farmed species that belong to different trophic levels in the food chain. In an IMTA system fish are farmed together with other species including bivalves (such as mussels and oysters), echinoderms (such as sea urchins) and algae or seaweed creating a more efficient, cleaner and less wasteful production system. IMTA allows nutrients from fish farms that are otherwise lost to the environment to be turned into useful products as they are utilized by these additionally grown species. IMTA is considered as one of the solutions to concerns about the future sustainability of aquaculture by increasing productivity and profitability while also reducing waste and over-reliance on raw materials from wild fish stocks.

The FP7 European research project IDREEM (Increasing Industrial Resource Efficiency in European Mariculture) aims to develop and demonstrate such IMTA technologies protecting the long-term sustainability of European aquaculture. The IDREEM project aims to demonstrate the benefits of IMTA through pilot commercial-scale testing, field research and modelling in collaboration with seven European small and medium enterprises (SMEs) paired to local Research and Technology Development (RTD) institutes (SME/RTD pairings). Interdisciplinary research within IDREEM will examine the obstacles and risks of using IMTA systems and apply and develop tools to overcome these constraints, whether they are economic, environmental, technical, social or regulatory. One of the tools applied is Life Cycle Assessment (LCA), which we use for answering two questions: 1) what are the environmental trade-offs for SMEs considering to move from monoculture aquaculture practice towards IMTA; and 2) what are the environmental trade-offs comparing IMTA species with their conventional monoculture alternatives.

The LCA work in IDREEM started with an LCA training of the different SME/RTD pairings. The goal of this workshop was to get the members of the pairings familiar with the main concepts of LCA and to familiarize them with the data collection needed for the LCAs. Following the LCA training the goal and scope definition and inventory analysis were started up. As a first step in the data collection, the present

finfish productive systems (monoculture, baseline or before IMTA systems) of all SMEs were described qualitatively. This included describing all relevant unit processes of each SME. As a second step, quantification of the inputs and outputs of the unit processes of the monoculture systems is currently taking place.

This paper presents first the methodology used to describe the SMEs monoculture and IMTA systems and second our first conceptual and qualitative results addressing the relation between questions posed, type of analysis needed and levels of integration achieved in IMTA. In the next 2 years of the IDREEM project quantitative LCA results will be produced and hopefully used in combination with the conceptual outcomes of this paper.

2. Methods

We started by qualitatively describing the current monoculture SME systems. As a first step in the qualitative description of the monoculture systems and keeping in mind the future comparison with the IMTA systems, the decision of assessing the SMEs monoculture systems in a detailed manner was made. The intention of this decision was to have as much data as possible available in an early stage of the project, including the SMEs' productive processes and activities, their material inputs and outputs and some ideas about their supply chain and waste treatment. A detailed description allows for a future aggregation while a "black box" type of description does not allow for disaggregation anymore neither for a detailed data collection in a later stage of the project. A detailed description was also expected to enable a better understanding of the true integration level of the IMTA species into the existing monoculture systems. Although the aim is to be as detailed as possible in the data collection the feasibility of this, of course, still depends on the availability of data within the SMEs.

The right part of Figure 1 shows a detailed scheme of an aquaculture production system, some general inputs and outputs but without showing the supply chain of materials for clarity of the graph. All inputs to individual processes are supplied by another productive system (upstream processes) or extracted from nature (environmental inputs). The left part of Figure 1 shows the "black box" aggregated type of description of the same system. The black box description is a simplified alternative to describe the monoculture systems, in which the material flows within the SME are not described.

As a second step in the process of qualitative data collection, pairings listed the activities (unit processes) performed and outsourced by each of them for their current monoculture production. A description of the practices used in each of the processes was also described as this helped in defining material inputs and outputs. The list of unit processes provided by each SME constitutes the foreground processes, for which data (primary data) will be collected by the pairings. All other economic activities in the supply chain of the SMEs i.e. upstream supply processes or downstream waste treatment processes will be described using literature or other secondary data sources. The monoculture LCAs consist of attributional cradle-to-gate analyses and do not include consumption of fish as part of the downstream processes. The boundary of the monoculture LCAs is thus at the gate of the SMEs, which depending on the activities of each SME, can be different. The right panel of Figure 1 shows common unit processes for all the SMEs, as well as possible boundaries and generic inputs and outputs to the monoculture productive processes.

Additionally to the monoculture activities, the pairings also described their preliminary plans for their IMTA systems. With respect to the level of integration of the planned IMTA systems (i.e. production activities and infrastructure) into the current monoculture systems, the proposed IMTAs are likely to move between two extremes.

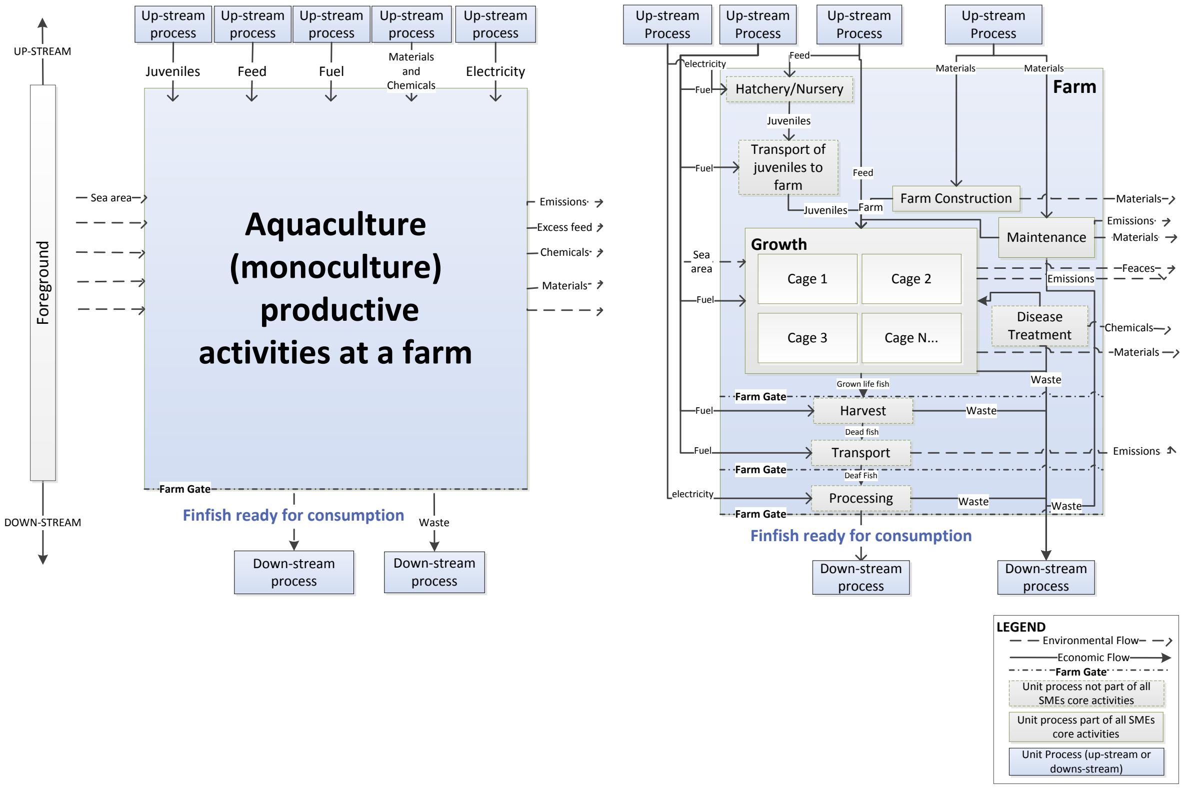

Figure 1. Left: "Black Box" aggregated and Right: Detailed disaggregated description of aquaculture farms

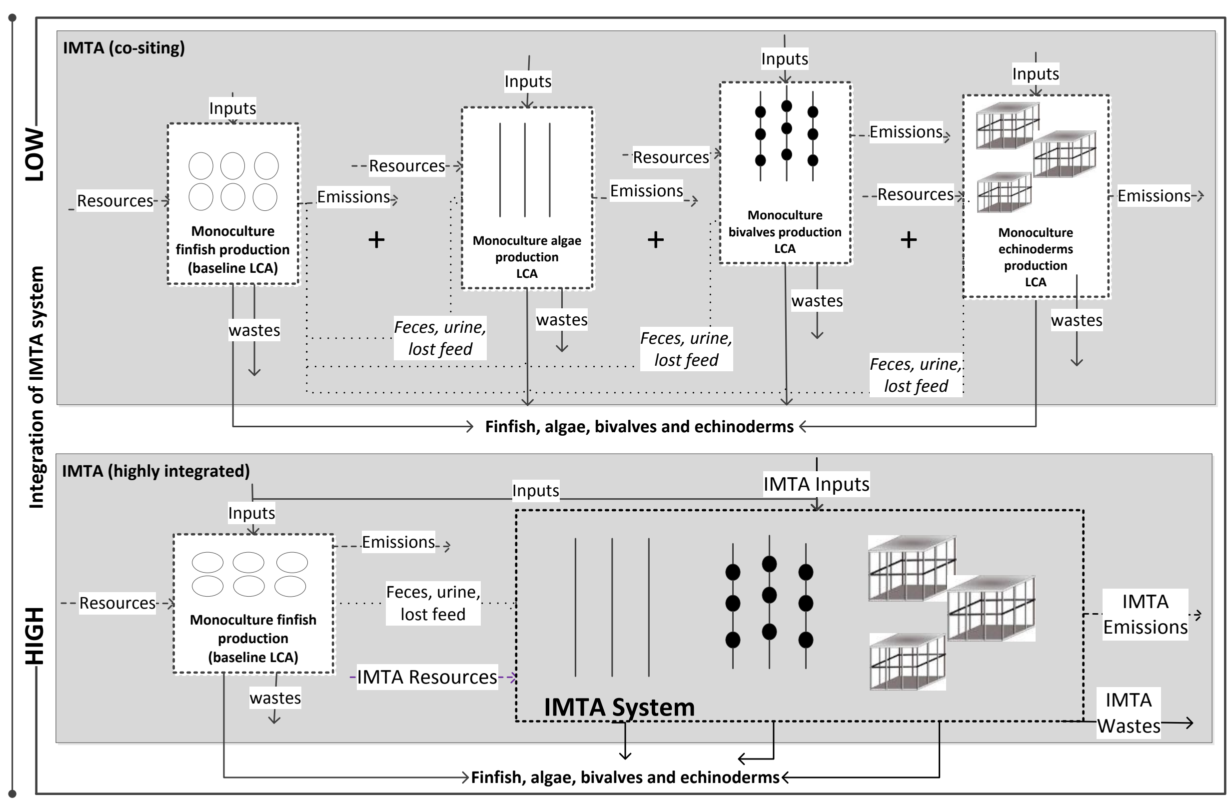

Figure 2. Level of integration of IMTA systems into the current production

On one side the IMTA system consists of a small add-on infrastructure and the operating activities (feeding, harvesting and related transport) are *highly integrated* into the current productive activities of the monoculture. In this case the IMTA system can be considered as highly integrated into the monoculture production system, and we refer to this extreme as the highly integrated type of IMTA.

On the other side the IMTA system corresponds to a relatively larger add-on infrastructure at sea constituting completely separate monoculture production lines for each co-cultured species, with its own inputs, resources, emissions and waste production. The latter extreme follows the description of IMTA by (Reid et al., 2009): "*Integrated aquaculture allows intensive management of several monocultures from different trophic levels within the same system, all connected by nutrient transfer through water"*. We refer to this extreme as the *co-siting* (or i*ndustrial symbiosis*) type of IMTA. We considered this extreme as *the low integration* extreme as only nutrient exchange in the water relates the different monocultures for the IMTA species.

Figure 2 represents schematically the two extreme levels of integration distinguished above: the upper part represents the lowest level of integration or co-siting (or industrial symbiosis) type of IMTA, the bottom part represents the highest level of integration.

Based on the descriptions of monoculture and IMTA systems, we now present our first conceptual and qualitative results addressing the relation between questions posed, type of analysis needed and levels of integration achieved in IMTA. These conceptual results consist of the scoping framework we intent to use further in the project once quantitative data becomes available for both monoculture and IMTA systems, in order to answer questions about the sustainability of IMTA as an environmental management concept that can minimize the environmental impacts of conventional monoculture aquaculture farms.

3. Results

In the methodology, we described how the monoculture and IMTA systems were qualitatively described by the pairings. Data to complete full attributional LCAs of the monoculture systems, are under collection by the parings and the IMTA systems are currently under pilot testing and are expected to achieve different levels of integration within the monoculture production activities and infrastructure. LCA will be used to answer two questions related to these systems: 1) what are the environmental trade-offs for SMEs considering to move from monoculture aquaculture practice towards IMTA (Q1: SME perspective); and 2) How do the environmental performance of IMTA species compare with their monoculture conventional alternatives (Q2: Product perspective).

Since the question determines the appropriate analysis and not the other way around, we present here the relation between these two questions, the type of analysis we will apply and the levels of integration achieved in IMTA systems. . We do so in order to determine the environmental trade-offs between the two systems that depend basically of three parameters: (1) the produce i.e. IMTA systems produce new co-cultured species next to finfish, (2) the nutrient discharge in the sea water due to lost feed and fish urine and feces, and (3) the life cycle impacts of the IMTA infrastructure e.g. upstream processes to produce inputs required to build and run the production of co-cultured species.

The flow diagram in Figure 3 shows the scoping framework for both Q1 and Q2. Q1 is a question from a company-perspective primarily, and is the main question for IDREEM. Companies are interested in increasing their eco-efficiency by cultivating economic species that grow on the nutrient discharge from the monoculture. This question is mainly addressed by a difference analysis which compares the environmental performance of the before and after IMTA system, taking into account differences in produce. In a difference analysis (Guinée et al., 2002) parts of the life cycle of the compared systems are expected to be quantitatively and qualitatively identical and are therefore omitted from the analysis to simplify it. The difference analysis can be performed for both extremes of integration and is thus independent of the level of integration as shown in Figure 3. However, the level of integration determines the magnitude of the trade-offs as in a low integration IMTA system the life cycle impacts of the IMTA infrastructure is expected to increase while in the high integration is expected to increase or perhaps decrease if inputs for the monoculture production are also used in the production of IMTA co-cultured species.

Moreover, full single-species LCAs can also be used to answer Q1. In this case, full attributional LCAs would be quantified for each monoculture species for all the co-cultured species in the IMTA systems and compared with the performance of the IMTA systems. Nonetheless, this would imply a change in the perspective for the question asked about the systems. In fact, we consider that full single-species LCAs leads to answer Q2, a question from a product-perspective, primarily comparing two ways of producing the same products (e.g., finfish, echinoderms, bivalves, algae, …): integrated in an IMTA system or by separate monoculture systems. In this case we will need to draft full single-species LCAs for each of the co-cultured species and compare these to the IMTA production of the same combination of species.

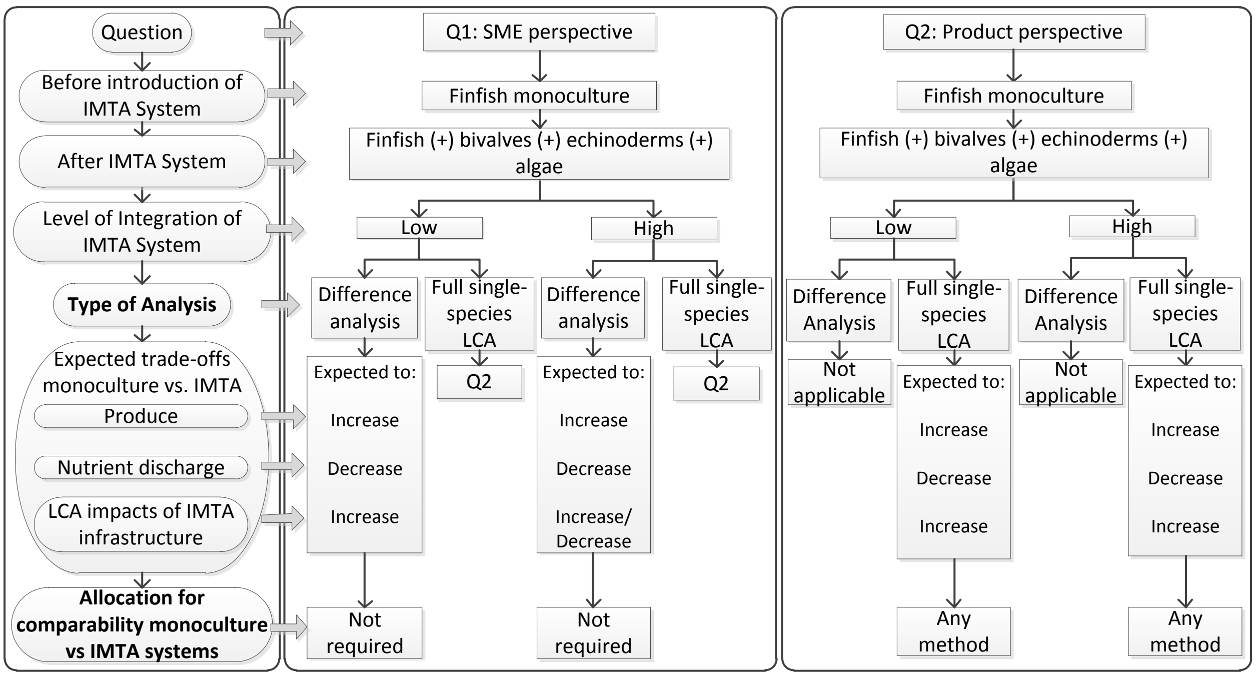

Figure 3. Scoping the IMTA systems LCAs starting from the question asked

The IDREEM SMEs currently produce only one species in their monoculture systems; as a consequence, we don't have a complete set of representative monoculture systems to compare the IMTA with. We will thus have to adopt existing LCA studies on these co-cultured species, adapt them to the SME conditions where possible, and take their results as second-best baseline system. The choice for separate, full single-species LCAs is again independent of the level of integration, but the more integrated the more challenging the allocation issues for the LCAs performed will become. To make the before and after IMTA systems comparable allocations is therefore required and any method, among which partitioning principles, substitution or system expansion should be applied. For Q2 we consider that difference analysis for any level of integration is not applicable as it would be very difficult to identify the equal parts of the before and after IMTA systems, and therefore full single-species LCAs would be the appropriate way to analyze such question.

4. Discussion

Our preliminary conceptual results show how questions determine the type of analysis to be applied. More particularly, the questions determine the scope of the LCAs to be performed, being a difference analysis omitting large parts of the systems compared or being a set of single, full LCAs for each species cultured. Focusing on the two starting question formulated different types of analysis can be used in order to calculate the life cycle environmental impacts of the IMTA systems implemented in the IDREEM project. The conceptual results also show that the type of analysis is independent of the level of integration of new species into the current monoculture system. Nevertheless, the level of integration of the productive activities of the co-cultured species (e.g., bivalves, echinoderms and algae) into the existing finfish monoculture system is expected to determine the magnitude of the trade-offs. The LCAs to be performed and the IMTA systems proposed by the involved SMEs with hopefully differences in levels of integrations will determine whether this expectation is correct or not. A complicating issue then is that the IMTA systems developed and tested within the IDREEM project will most likely not be in a stage beyond pilot-scale. This will be challenging for the LCA work as it implies that the LCAs can only deal with the pilot-scale on a more or less empirical data level. We will have to apply scenario and up-scaling techniques to estimate the LCA results for a full commercial exploitation of the IMTA plans.

5. Conclusion

IMTA is regarded as an environmental management concept that can minimize the environmental impacts of conventional monoculture aquaculture farms while expanding their economic base (Price & Morris, 2013). These characteristics of IMTA systems hold as long as 1) there are no significant trade-offs between the discharge reduction and the life cycle environmental impacts of the IMTA system and 2) the species to be used in the IMTA system are potentially commercial species produced at scales that yield sufficient additional profit.

In this paper we explored the relations between questions asked, levels of integration foreseen for the IMTA systems and the type of analysis to be performed to evaluate the environmental trade-offs between monoculture and IMTA systems, particularly those part of the EU FP7 IDREEM project. Our preliminary conceptual results show a scoping framework with some of the expected trade-offs and also shows that although the type of analysis to be used is independent of the level of integration of new species into the current monoculture system, the magnitude of the environmental trade-offs between the discharge reduction and the life cycle environmental impacts of the IMTA system can depend on the level of integration.

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Building consensus for assessing land use impacts on biodiversity in LCA

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ABSTRACT

The UNEP SETAC Life Cycle Initiative has successfully provided a platform for consensus finding in the area of environmental indicators. A flagship project aims to provide global guidance and consensus on a few environmental indicators, including land use impacts on biodiversity. Land use and particularly land use change is one of the main drivers of biodiversity loss. However, no consensus exists yet on which indicators to use to quantify land use impacts on biodiversity within LCA. This is challenging for LCA on land-based products and production systems based in extensive land use, such as agriculture, forestry, open mining etc. The flagship project evaluates existing methods and models based on their ecological relevance, scientific robustness, transparency, reproducibility and applicability, among other criteria. The project is in progress with the ambition of delivering recommendations in 2015. This presentation focuses on identified methods and models to be included and review criteria for evaluation.

Keywords: Life Cycle Analysis, Life Cycle Impact Assessment, Biodiversity, Land Use, Ecological Indicators

1. Introduction

Environmental impacts of production and consumption are increasingly coming into the focus of companies and governments. Life Cycle Assessment (LCA) is one of the most important approaches to quantify environmental impacts of products from cradle to grave, in which the product system, process or service life cycle is mapped and environmental impacts are quantified. Applied as a decision-making tool and/or as a support to policy development, it makes use of indicators to consider different environmental aspects over a system's life cycle. Environmental indicators, i.e., quantified representations of impacts associated with specific impact categories, are required to measure the greening of the economy and of products. With increasing economic globalization, there has been a steadily growing need to create a worldwide consensus set of environmental indicators.

The UNEP/SETAC Life Cycle Initiative successfully provided the platform for consensus finding in the area of environmental indicators (USETox covering human and ecotoxicity) and LCA databases (Shonan Guidance Principles, UNEP/SETAC 2011). The work builds on previous experiences including the work on Product and Organization Environmental Footprints (PEF/OEF) of the European Commission and of the Water and Land Use working groups of the Life Cycle Initiative. A flagship project has now been established by the Life Cycle Initiative intended to run a global process aiming at global guidance and consensus building on a limited number of environmental indicators, including indicators those for assessing impacts from land use (LU) interventions, such as land use change (LUC), on biodiversity. The mandate of the taskforces started in early 2014 and will terminate in late 2015.

Biodiversity ranks high among the indicators of interest for the Initiative. An intensive transformation of natural areas over the past 50 years, mainly driven by economic growth and agricultural intensification and supported by technological development, have caused the loss of important environmental assets (MEA 2005). Land has been used for the production of goods and services along a global network of supply chains, often involving different temporal and spatial scales and various environmental pressures (UNEP 2012), which are mostly difficult to account for (Yu et al. 2013). These range from impacts on provisioning ecosystem services, with economic interest to humans, to the loss nature's intrinsic values and natural assets, such as biological diversity.

The increasing rates of land conversion in the last decades have led to a stronger engagement of different actors to enhance the sustainability of commodity supply chains (Newton et al. 2013) and the need to develop and implement tools to better understand the influence of supply chains on land use problems. In this context, the analysis of interconnected chains and linkage with specific policy are key steps on the assessment of impacts. Considering the manufacturing and production activities as some of the main drivers to land use and land use change, there is the need to focus on the prediction of potential impacts, such as biodiversity loss. There is also the need to assign responsibility for damages spread out across the supply chain, instead of focusing solely on final product providers.

However, despite these substantial contributions to address this issue in LCA, no clear consensus exists yet on the use of (a) specific impact indicator(s) to quantify land use impacts on biodiversity within LCA. This lack of agreement not only limits the application of correlated models, but also imposes constraints on the comparability of results of different studies evaluating land use impacts.

In this paper we establish the operational principles of the flagship project regarding biodiversity impacts from LU and LUC. We show how the taskforce in charge will assess current impact assessment methodologies and the process that will take place towards widespread consensus. We present briefly some preliminary and expected results and deliverables before the end of the mandate of the taskforce in late 2015.

2. Methods

2.1. Goal and scope of the taskforce

The objective of the taskforce is to conduct an overarching process aiming at global guidance and consensus regarding indicators and methods for the assessment of biodiversity impacts from LU (and, if possible, LUC) in LCA. An agreement will be pursued among experts, practitioners and stakeholders as to which methods respect a pre-established set of requisites. It is the task of the group to steer this process and collect all feedback in order to establish what are those requisites and in what cases, if any, those requisites are not met. It is beyond the scope of this taskforce to develop new methodologies, but it is expected that during its work the main aspects to tackle in future developments will be identified. Also, it might be suggested to combine existing proposals or to adapt and integrate existing methods from outside the LCA field (e.g. from recent research in ecology, conservation or other environmental sciences dealing with land use).

2.2. Participants and Process

To succeed in this task, this global effort involves world leading environmental and LCA scientists to develop scientifically robust indicators suitable for a global consensus. The taskforce includes not only experts in the LCA field, but also biologists, ecologists and experts in other disciplines relevant to the task.

The taskforce members contribute as part of one of two levels of engagement, namely:

- *Full members*, who have an active role in composing the deliverables. There are approximately 10 full members at present.
- *Agenda members*, who monitor the taskforce meetings and internal documents, providing comments when necessary. There are approximately 20 agenda members at present.
- *Stakeholders and interested parties*, who will lend their expertise to the group during specific stages (to be determined).

The workflow of the taskforce will take place according to the ensuing steps:

1. Identify domain experts, particularly beyond the LCA community, in order to engage with the top landscape ecologists and biodiversity scientists for a truly robust and internationally recognized indicator. Domain experts may engage directly with the taskforce or participate solely in consensus-building workshops.
2. Determine the relevant evaluation criteria for models of biodiversity change in LCA.
3. Identify impact indicators / models for land use impacts on biodiversity.
4. Assess models against the evaluation criteria, which may be expanded to include biodiversity-specific criteria deemed relevant during the assessment stage.
5. Collect feedback at the SETAC Europe Annual Meeting 2014 (http://basel.setac.eu/?contentid=636), an LCIA workshop, where the taskforce will share results of the method comparisons as well as the analysis and selection of models or model elements to represent the consensus.
6. Frame meetings (expected in autumn-winter 2014-15) to obtain inputs from specialized domain experts on the proposed assessment framework and indicators.
7. Build consensus and refine approaches during year 2, when the taskforce will prepare a report proposing the assessment framework, as well as the selected models and factors, as an input to a Pellston workshop.
8. Hold a Pellston or other technical workshop in 2015 for participants to analyze the inputs from the taskforce, establishing consensus on impact assessment approaches and factors.
9. Make recommendations and publish these in the scientific journals, as a follow-up from the technical workshop.

2.3. Deliverables and expected impact

The flagship project will deliver a global guidance publication with a supporting web system that may include ready to use datasets (impact assessment factors) for download. The publication will also include a critical review of the existing framework for land use impact assessment in LCA, accentuating key elements and shortcomings; plus a brief review of existing indicators and guidance on how to best establish regional impact indicators. In case global consensus on environmental indicators cannot be achieved, the publication will focus on crucial developments needed in order to steer impact assessment methods developers in the right direction.

These deliverables will confer international consensus and credibility to the quantification of environmental impacts on biodiversity generated by production systems in the context of e.g. product environmental footprinting, eco-labelling, etc. It is expected that the results and recommendations of this flagship project will be eventually incorporated into international and national initiatives in these areas.

The taskforce will also try to reinforce its mission by attempting to find connections with other existing similar initiatives. The consensus-finding process will end with a SETAC Pellston workshop, a one week workshop event in a remote location where invited experts and stakeholders discuss and agree on the recommended environmental indicators.

3. Preliminary results and discussion

3.1. Impact pathways

The assessment of LCA models is a necessary step before trying to build consensus in the community. As stated in point 2 of Section 2.2, to set up this assessment, one needs to commit to a set of principles to which the evaluation must conform.

The taskforce initiated this process by revisiting the impact pathways, originally proposed by Koellner et al. (2013a) that lead from LU and LUC to changes in the state of biodiversity. We completed the scheme with a comprehensive description of the main drivers of change and their consequences to variations in biodiversity indicators such as species richness and also ecosystem services provided by biodiversity. The result is shown in Figure 1. Impact assessment models will then be analyzed by assessing the inclusiveness of the pathways described in the figure.

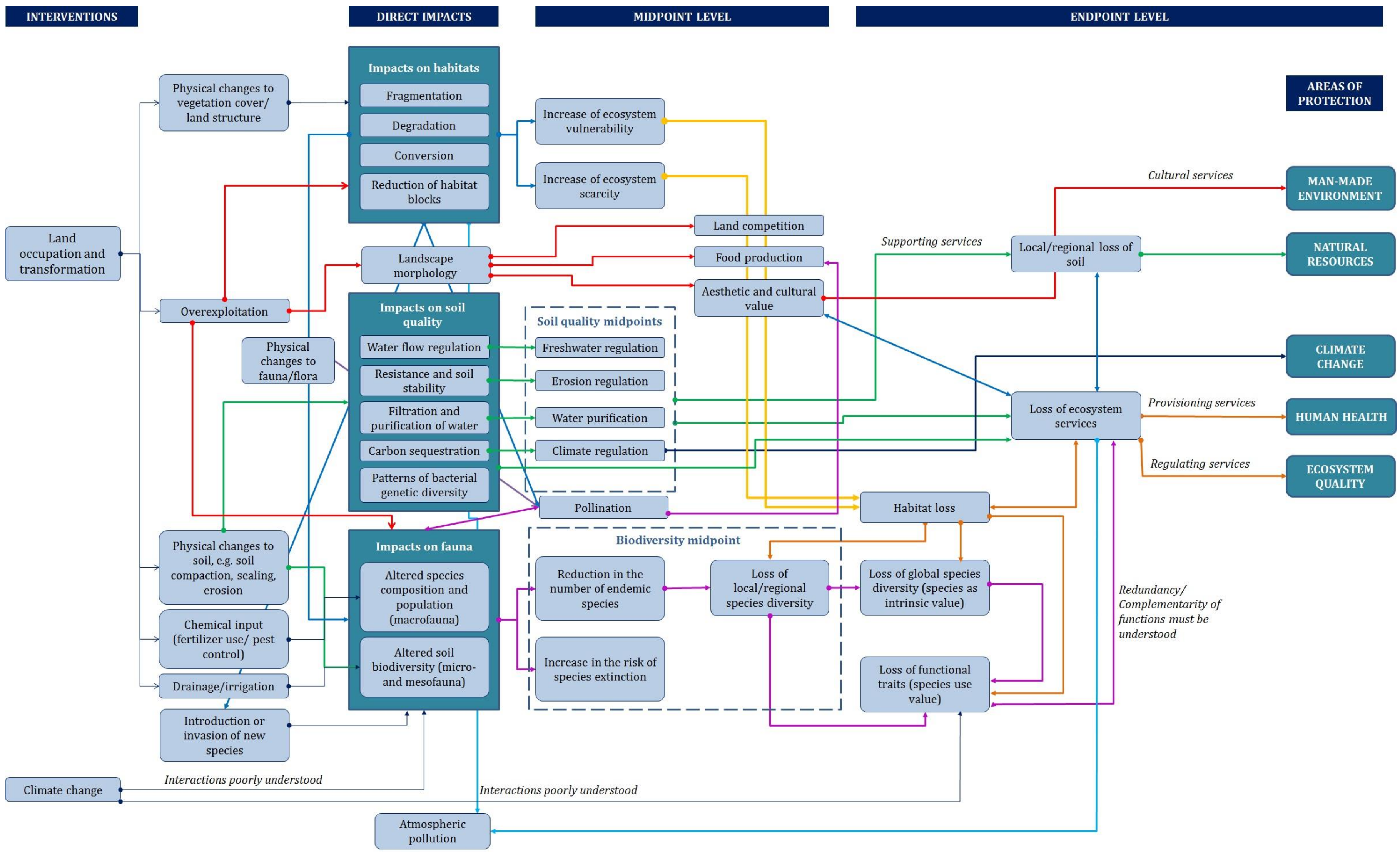

Figure 1. Major impact pathways leading from land use and land use change to changes in the state of biodiversity, and feedback loops that change the state of biodiversity and reinforce / counterbalance early drivers (adapted from Koellner et al. 2013a).

3.2. Review of LCA methods for biodiversity

After the main cause-effect pathways are determined, the taskforce reviews the methods for biodiversity assessment. While this work is still ongoing, some preliminary notes can be drawn from the effort.

Modelling efforts have yielded significant progress in the last two decades (Lindeijer 2000, Weidema and Lindeijer 2001, Brentrup et al. 2002, Koellner and Scholz 2007, Milà i Canals et al. 2007, Michelsen 2008, Schmidt 2008, Penman et al. 2010, de Baan et al. 2013a, de Baan et al. 2013b, Koellner et al. 2013a, Souza et al. 2013, Coelho and Michelsen 2014). These studies follow a systematic procedure that can be systematized in Figure 2.

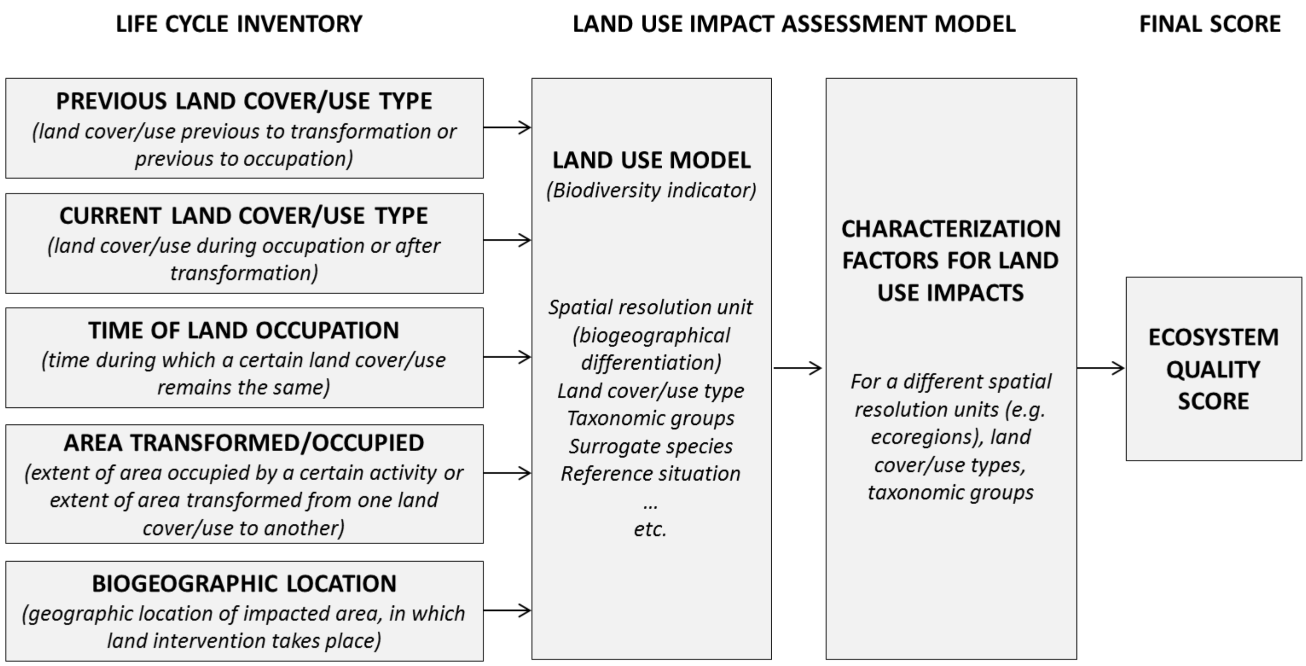

Figure 2. Key elements currently employed on the assessment of land use impacts in LCIA.

The group will extend the scope for identifying models, ideally including models from ecology that may be a good match for LCA even if they currently require some adaptation to fit in the scheme presented in Figure 2.

3.3. Systematic assessment of methods

The taskforce group also developed a systematic evaluation matrix based on the assessment of land use impact indicators performed for the ILCD in 2010. The current version of this matrix includes as relevant categories of assessment, for example, the following evaluation categories:

- *General completeness of scope*. What is the geographic and temporal scope of the assessment offered by the method? What is the reference state and the nature of the indicators (marginal or average impacts)? What is the taxonomic coverage? Are permanent impacts considered and double counting of impacts from different drivers avoided?
- *Compatibility and availability*. Is the method compatible with the established architecture of the overall LCA framework? If not, what would be required to make the method compatible? Are the underlying data and impact factors available for use by LCA practitioners?
- *Environmental relevance*. Does the method include the major impact pathways linking LU and LUC to biodiversity loss? Are the compositional, structural and functional attributes of biodiversity reflected in the midpoint and endpoint indicators? Are impacts across components (genes, species, ecosystems) reflected in the method? Is there a specific link to threatened or rare species and ecosystems?

- *Scientific robustness & certainty*. Are uncertainties quantified and presented in the method documentation? What recommendations can be made to deal with these uncertainties when assessing LU and LUC impacts? The taskforce's mandate includes the involvement of stakeholders, among which are policy-makers and other final users of the results obtained using the method(s) chosen. One crucial aspect is thus determining the usability of the indicator(s) that the group will recommend. This indicator must comply will all scientific requirements, and also provide useful information that practitioners can clearly interpret without being an overburden in the interpretation stage.

Tables 1 and 2 show selected evaluation criteria for the categories of "Completeness of scope" and "Environmental relevance", respectively. Descriptions of the type of questions and specific criteria to check against each method are shown in the third column.

Theme	Evaluation criterion	Description
1. General completeness of scope		
Indicator(s) and model(s)	Marginal (M) or average (A) damage, or not defined (ND)	Is the model based on average damage (A), marginal increase in damage (M), or is it not defined (ND)?
	Cause-effect relationships	The model based on a) best estimate or b) on precautionary cause-effect relationships
	Reference state	What is the reference state assumed by the method? Potential Natural Vegetation (PNV), Historical Reference Vegetation (Historic) or Average Current Vegetation (Average) or a mix depending on the context (Mix)
	Double counting	How are double counting problems when using empirical data addressed? Are they avoided, or not even mentioned? Describe any potential double counting (e.g. common impact attributed twice to interacting inventory flows causing joint effect).
	Number and description of land use/cover classes	Total number of individual land use/cover classes covered with specific characterization factors provided. Score system is based on reference to 33 land use/cover classes listed in Koellner et al. (2012) as a desirable goal, aggregated at a second order of detail (i.e. 1.2.1 = Forest, used, extensive). These are in turn drawn from EcoInvent, Alkemade et al. (2009) and Koellner and Scholz (2007a)
	Intensive/extensive LU class distinction	Do the impact/characterization factors differentiate between intensive and extensive land use classes? In terms of high/low-input agriculture, intensive/extensive pasture, clear-cut or selective forestry? Does it consider the cultural value of landscapes?
	Spillover effect (immigration credit and extinction debt)	Is the spillover effect between reference or original habitat dealt with in the method (or even assessed)? Immigration credit expresses a spatial effect of individual colonizing from nearby natural habitat. Extinction debt and immigration credit involve a time lag in biodiversity loss following conversion. Immigration credit could be, e.g., assessed using a proxy of distance to source habitat, extinction debt using a proxy of time since transformation.
Spatial and temporal characterization	Spatially and temporally explicit characterization	How can the method be applied to another spatial and temporal context? Is the characterization model adaptable to spatially and temporally explicit evaluation, and the input data needed clearly communicated?
	Permanent impacts	Does the method quantify irreversible, permanent impacts (species extinction, biotic or abiotic thresholds such as compaction, topsoil loss, extreme habitat isolation)? How are permanent impacts considered? Using what models and assumptions? Considering which scale?
	Geographic scope	For which geographic scope (regions) are the model and factors determined? What is the origin or scope of the data on which the method is based? List specific locations, single countries (list names), realms (list realm, AA: Australasia, AT: Afrotropic, IM: Indomalay, OC: Oceania, PA: Palaeoarctic, NA: Nearctic, NT: Neotropic), other (specify). Realms based on Olson et al. (2001).
	Spatial resolution	What is the level of spatial resolution of the method (generic/global, realm/continent, national, sub-national, pixel-based)?
	Taxonomic coverage	What is the taxonomic coverage of the method, in terms of the species groups used to develop the indicator: Soil fauna (SF), Herbaceous Plants (HP), Woody Plants (WP), Mosses (Mos), Lichens (Lic), Fungi (Fun), Mammals (Mam), Birds (Bir), Amphibians (Amp), Reptiles (Rep), Arthropods (Art) and other Invertebrates (Inv)

Table 1. Working version of selected criteria included in the method evaluation canvas shown for the category of "*General completeness of scope*". Adapted from ILCD land use method assessment 2010.

Theme	Evaluation criterion	Description
2. Environmental relevance		
General properties	Underlying biodiversity model(s)	Is there a specific underlying biodiversity model used (e.g. species area relationship, habitat suitability models)? Or are values based on an empirical pattern e.g. meta-study)?
	Land transformation framework	How is land transformation considered? E.g. LCIA framework: area x impact x recovery time
	Land occupation framework	How is land occupation considered? E.g. LCIA framework: area x impact x occupation time
Hierarchical components considered by method	Genetic level	Genetic indicators (e.g. allelic diversity, phylogenetic diversity)
	Single-species population level	Population indicators (e.g. population reductions, individually for species or averaged across species, or red list index of threatened species)
	Multi-speicies community level	Community indicators (e.g. richness loss or compositional change)
	Landscape/ecosystem level	Ecosystem indicators (e.g. habitat loss, SAR-based loss, fragmentation, isolation etc.)
Biological attributes considered by method	Composition diversity	Compositional indicators (e.g. relative abundance, presence and relative proportion of biodiversity features)
	Functional diversity	Functional indicators (e.g. ecological processes, nutrient turnover, immigration/emigration, NPP, functional group diversity)
	Structural diversity	Structural indicators (e.g. configuration of elements in space, habitat structure metrics, foliage density, topography, number of vegetation layers etc.)
Conservation relevance of indicator(s)	Quantitative change in species extinction risk	Does indicator measure a quantitative change in extinction risk?
	Vulnerable/red-list or rare/endemic species treated separately	Does the metric take into account threaten status, rare, endemic species distinctly (describe)
	General assemblage (all species) considered (e.g. richness, similarity)	Does it consider only (/in addition) overall assemblage change (i.e. not differentiating between species)?
	Vulnerable/red-list or rare/endemic ecosystems treated separately	Does the metric treat red list, threatened, rare, endemic ecosystems distinctly (describe)
Specific cause-effect pathways via interventions 1...N	Intervention 1: Ecosystem fragmentation and loss Intervention 2: Local vegetation cover modification Intervention 3: Landscape morphology change ...Intervention N	Which cause-effect chains for biodiversity loss are included with acceptable quality? Each environmental intervention has a corresponding cause-effect chain leading to direct impacts on biodiversity. Which indicators are used to assess impacts along the pathway?

Table 2. Working version of selected criteria included in the method evaluation canvas shown for the category of "*Environmental relevance*". Adapted from ILCD land use method assessment 2010.

3.4. Main questions to be addressed

The task force builds upon some key publications (Milà i Canals et al. 2007; Curran et al. 2011; Koellner et al. 2013a; Koellner et al. 2013b) that have experienced strong consensus among experts. These relevant publications, together with the work carried out by the group, allowed the taskforce to draw as early as the first trimester of work some intuitions about what particular aspects of the methodologies are stronger candidates for consensus and which will need more attention. These aspects, summarized next, will likely be the focus in the application of the evaluation criteria.

Three basic areas where there exists some form of preliminary implicit agreement can be highlighted:

- *Key elements in the assessment of land use impacts in LCA*. Occupation (area x time) and transformation (area) are the so called environmental interventions leading to land use impacts. In the current framework, the difference between land quality maintained (occupation) by the production system and a reference situation integrated over time and area is the measure of the impact. The measure of land quality requires agreed impact indicators (Figure 3).

- *Geographical differentiation*. The assessment requires some degree of spatial resolution.
- *Definition of land cover/use types*. A minimum distinction between land use/cover types is required (Koellner et al. 2013b).

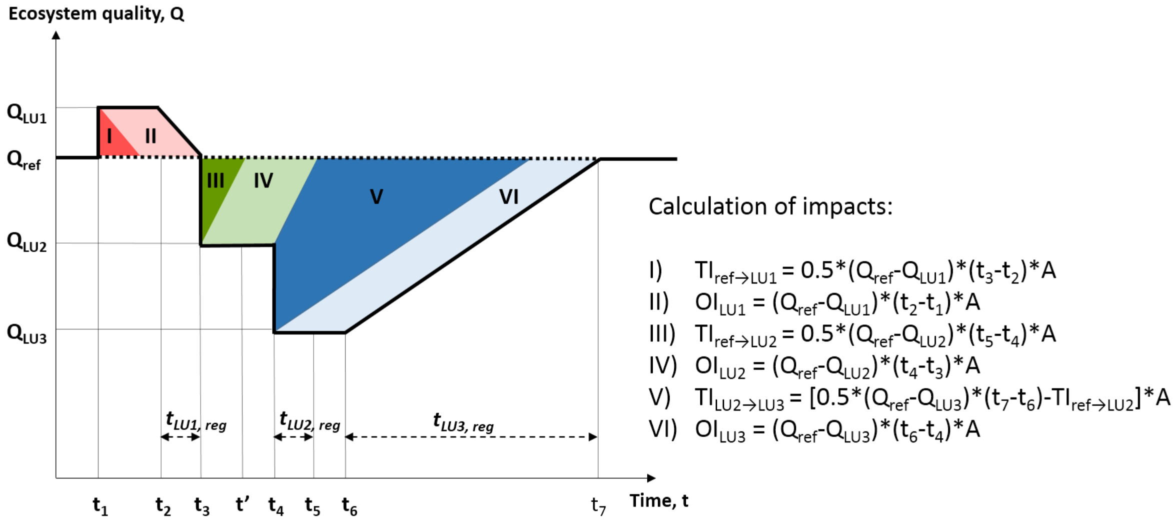

Figure 3. Simplified illustration of transformation impact (TI) and occupation impact (OI) for three land use types with different regeneration rates (tLUI, reg; tLU2, reg; and tLU3, reg). For simplicity, the area A of occupation and transformation, which would embrace the third dimension, is not shown in the graph, but in the equations.

The topics mentioned above will nevertheless be part of the review and eventually (if necessary) the group may recommend improved versions to the way they have thus far been operationalized in LCA.

Similarly, key areas of disagreement or generalized lack of satisfaction with the methods can be identified early and will be taken with care during the consensus-building exercise. These areas include, but may not be limited to:

- *Generic framework of land use impact assessment (Figure 3).* Although the framework in its abstract formulation is subject to widespread agreement, there are aspects in its operationalization that require careful consideration. As examples, there are several options for a reference state: historic, potential natural vegetation, natural climax, present state or others. It is also unrealistic to assume a null transformation time for most land use conversions. The taskforce will consider if it is acceptable, to facilitate consensus, to limit the assessment to occupation and exclude transformation impacts. The group will also equate the consequences of assuming full recovery vs. consideration of irreversibility of damages. We use this issue as an example of the analysis the group is currently performing in section 3.5.
- *Indicator(s) of biodiversity.* The taskforce will consider how best to represent biodiversity – as a single indicator (and in that case which) or in multi-indicator form; this is the main focus area for the task force. Species richness is commonly taken as a proxy to the state of biodiversity, but aspects such as species evenness or species functionality are equally crucial, as well as indicators at different levels (genetic diversity, population dynamics, ecosystem structure, vulnerability or scarcity, among others). Other types of indicators such as structural or pressure indicators are possible and will be assessed. Linked with the indicator(s) chosen, the group will define a set of ecological models used. If the strongest models involve species data, the group will attempt to assess which taxonomic groups should and can feasibly be included in the analysis, and assess current methodologies in face of their assumptions and data availability.
- *Spatial resolution.* The group will consider the scale at which impacts should be measured during impact assessment modelling. Since many ecological models are sensitive to data, it is not a trivial

issue to interpolate country-scale characterization factors from local ones and vice-versa, for example.

- *Impact assessment data requirements*. Linked to on-going data collection / reporting efforts, the taskforce will balance its thoroughness in terms of exhausting the ecological underpinnings of biodiversity change from land use and land use change with what is realistic to implement in LCA. Data limitations, as well as inventory limitations, together with the fact that LCA strives to calculate *potential* impacts, will necessarily require the taskforce to target feasible rather than exact modelling accuracy.

3.5. Example: review of the current framework

Land use impacts in LCA are typically associated with two inventory flows, land occupation and transformation (Lindeijer 2000, Koellner 2003, Milà i Canals 2007), which lead to a change in a defined quality indicator, Q, over a certain time period of time t. This is illustrated by Figure 3.

One crucial variable in this framework is the reference state, since its definition is a key element in the calculation of impacts in the temporal-spatial model. Currently, the so called Potential Natural Vegetation (PNV), as introduced by Tüxen (1956), is used as the reference. PNV concerns the vegetation that would develop in the absence of any human influences and in equilibrium with climatic conditions (Chiarucci et al. 2010). In LCA, land cover/use types are defined by Koellner et al. (2013a).

However, the recovery process of land is dynamic and the species present in the final state may be different – not meaning better or worse - to the original state (Curran et al. 2013). It is also questionable whether the resulting indicator is useful to policy-makers or fair to present-day interventions. Comparisons with, for example, land uses in the recent past could be easier to interpret than comparisons with idealized situations (see e.g. Milà i Canals et al. 2013). Further, although the framework technically allows improvements of ecosystem quality after LUC, the comparison with an idealized semi-natural system normally yields a loss of quality in the majority of situations. The framework also has a hard time dealing with dynamic effects such as temporal summation of impacts, since the past history of the field will have an influence in the ecosystem resilience (e.g. a highly fragmented plot is more likely to see higher quality drops), and thresholds, namely disturbance levels from which the ecosystem is no longer able to recover. To consider these effects one needs to build into the model the concepts of ecosystem resilience and resistance, which is impossible if the scale of the disturbance and the environmental quality of the ecosystem are not measured. This insight is relevant to evaluate biodiversity models.

One of the expected outputs of the task force will be to translate relevant criticisms (such as the reliance on PNV as the baseline for determination of impacts) into recommendations. Using this example, the group identified the baseline of the modeling framework as a limitation and will study alternatives and their consequences. Alternatives to PNV may be country-specific biodiversity targets, a LU mix at a reference year (e.g. 2000), or others.

4. Conclusion

Land use and particularly land use change from natural habitats to cropland and other human land uses is one of the main drivers of biodiversity loss and degradation of a broad range of ecosystem services (Sala et al. 2000, MEA 2005, Lenzen et al. 2009), by means of habitat conversion and/or fragmentation. However, land competition is very likely to increase in the future (UNEP, 2014) and the task to halt biodiversity loss became one of the top global priority issues.

LCA has a decisive word to say because of its nature as an aggregator of direct and indirect impacts allows producers and policy-makers to assign responsibility for damages to biodiversity. Given the importance of the environmental pressures on biodiversity and how much LCA can potentially contribute to conservation targets, it is imperative to align the LCA community around points of consensus, as well as point the way forward to better models that can service the community. The UNEP/SETAC taskforce intends to give a determinant contribution to this shared goal.

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Multi-criteria decision analysis of feed formulation for laying hens

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ABSTRACT

Increase awareness of environmentally sustainable food products and sustainability reporting in the food industry led to assessment of environmental performance of agri-food products using life cycle assessment (LCA). Feed production is mainly responsible for the overall environmental impact of egg and broiler production. In addition, feed cost accounts for a significant portion of the total cost of egg production. Feed formulation is a complex process of quantifying the amount of feed ingredients to satisfy nutritional requirement of layers. Traditional linear programming models can find least cost combination of feed ingredients that meet nutritional requirements. However, it has limitations of rigidity that cannot solve several conflicting objectives simultaneously. Alternatively, multi-criteria decision analysis (MCDA) can be used to find the optimal combination of feed ingredients that meet nutritional requirements at the lowest possible cost with the minimum carbon footprint. The results suggest that the carbon footprint of feed formulation could be reduced with a modest increase in cost.

Keywords: life cycle assessment (LCA), multi-criteria decision analysis (MCDA), carbon footprint, cost, feed formulation

1. Introduction

Consumer awareness of environmentally sustainable food products and sustainability reporting are responsible for a growing trend in sustainability of the food supply chain. Consumers are more conscious about how foods are produced and their consequential impacts on the environment. Many stakeholders in the food supply chain including producers, processors and retailers are adopting sustainable practices to address consumer demand for environmental friendly products and communicate the environmental profile of their products. Increasingly food retailers are integrating sustainable considerations into their supply chain to reduce cost, to enhance corporate reputation and to differentiate their products from competitors. In response to the demand and need for sustainability reporting, a life cycle assessment has been recently completed to assess the environmental performance of egg production in Alberta.

Key findings of LCA results identified feed production as the main contributor to greenhouse gas emissions (carbon footprint) (70%) of egg production in Alberta. The results were consistent with other LCA studies of egg and broiler production that identified feed production as a major contributor of greenhouse gas emissions (46-80%) from egg and broiler chicken production (Pelletier 2008; Sonesson et al. 2009; Wiedemann and McGahan 2011; Wiedemann et al. 2012). In addition, feed cost accounts for a significant portion (46-54%) of total cost of egg production (Martin et al. 1998). Improvement in the feed formulation can help to optimize feed efficiency.

Feed formulation is a complex process of assembling a blend of feed ingredients that meet the nutritional requirements of layers. Inadequate nutrition may lead to a reduction in egg size and production. Energy, protein and amino acid levels are major factors in feed formulation to achieve the best performance of egg production (Elliot 2012). Traditional linear programming has been used to formulate the least cost combination of feed ingredients that satisfies a specific level of nutritional requirements of layers. However, it has limitations of rigid assumptions that cannot solve several conflicting objectives such as cost minimization and reduction of environmental impact (Castrodeza et al. 2005; Rehman and Romero 1984). Compared to the linear programming, multi-criteria decision analysis (MCDA) is a more flexible method to solve several conflicting objectives simultaneously. Using MCDA, this study builds on the LCA study results to find the best compromise solution to the optimal combination of feed ingredients that satisfies economic, nutritional and environmental criteria.

2. Methods

2.1. LCA of feed ingredients

Carbon footprints of feed ingredients were assessed using ISO standards 14040/14044 (2006). This study included crop production on the farm, the subsequent drying, processing and the transport of feed grain to the farm. The functional unit used was 1 kg of feed ingredient at the gate of the feed mill. Background processes associated with input production (manufactured fertilizers, pesticides, agricultural machinery, and diesel), field operations and the transport of feed grains to the feed mill were adapted from the ecoinvent database (Nemecek and Kagi 2007). The electricity grid mix value was modified using Alberta data for energy sources and electricity importation from other provinces.

Life cycle inventory data for wheat, barley, peas and canola were developed using Alberta crop production data. Two farming systems (conventional and no tillage systems) were modeled for the four crops. Crop yield (2009-2011) and the proportion of each tillage system were used from the Alberta Agriculture Financial Service Corporation (AFSC – crop insurance). Corn, corn dry distiller grains (DDGS), soybean and soymeal were adopted from ecoinvent database using U.S Midwest data. Attributional LCA modelling was applied to the study because the main purpose of the study was to measure the total greenhouse gas emissions of feed ingredients based on average production data and the effects of any changes in level of output were beyond the scope of the study.

Economic allocation was used to allocate environmental impact of the by-products of feed ingredients such as soymeal, canola meal and corn DDGS because physical relationships of feed materials differed significantly from one feed type to another depending on nutritional contents such as energy, protein and essential amino acid (FAO 2014). Moreover, the prices of feed ingredients reflected their nutritional value. In this case, economic allocation was the best possible option to allocate environmental impacts of feed ingredients and their by-products in a consistent manner and on the basis of meaningful relationship between nutritional values and prices (FAO 2014). Limestone, dicalcium phosphate and salt were used from the ecoinvent database. Vitamin premix was used from Nguyen et al. (2012).

Global warming potential was considered as a single environmental impact indicator to analyze in the model because global warming potential was widely used as a standalone impact indicator using internationally recognized greenhouse gas quantification protocols such as ISO/TS 14067 and PAS 2050 to communicate the results to the public through carbon labelling schemes. For life cycle impact assessment method, IPCC 2007 GWP 100a was selected from SimaPro 7.3.3 to evaluate the global warming potential of feed ingredients (Goedkoop et al. 2008; Goedkoop et al. 2010).

2.2. Multi-criteria decision analysis (MCDA) for feed formulation

Multi-criteria decision analysis can be used to find the best comprise solution among several conflicting objectives. In order to model multiple objective programming, it needs to find target values for each of objectives (Tozer and Stokes 2001). This study focuses on two objectives – cost minimization and reduction of carbon footprint. Cost target (C*) and impact target (I*) are solved using traditional linear programming. C* is determined by a LP model of cost minimization as follow:

$$\min C = \sum_{i=1}^{I} \pi_i X_i \qquad 1$$

Subject to:

$$\sum_{i=1}^{I} a_{ij} X_i \geq b_{ij} \ \forall \ j = 1,2,..., J-1 \qquad 2$$

$$\sum_{i=1}^{I} a_{ij} X_i \leq b_j \qquad 3$$

The objective function specified by equation 1 describes the summation of the prices of the i feed ingredients (π_i) multiplied by its amount (X_i) used in the optimal feed formulation. Equation 2 and 3 are recommended nutritional lower and upper bound constraints. The coefficients a_{ij} measure the amount of the jth nutrient in the ith feed ingredients and b_j limit the allowable minimum or maximum amount of the jth nutrient in the feed formulation depending on the sign of inequality. Crude protein (%), metabolizable energy (kcal/kg), available phosphorus (%), calcium (%) and essential amino acids such as methionine (%), methionine + cystine (%), lysine (%), threonine (%), tryptophan (%) and isoleucine (%), are considered as the jth nutrient according to diet specification from commercial poultry nutrition (Leeson et al. 2008). Wheat, barley, corn, corn DDGS, soymeal, canola meal and peas were included as major feed grain crops. Other feed ingredients such as limestone, canola oil, dicalcium phosphate, salt and vitamin premix were also included in the feed formulation based on recommended diet specifications (Leeson et al. 2008).

Diet specifications for layers (60-70 weeks) were used based on guidelines for nutritional requirements of layers from commercial poultry nutrition (Leeson et al. 2008). Nutritional values of feed ingredients were collected from commercial poultry nutrition by Leeson et al. (2008). Price data (2013) for wheat, barley, corn, corn DDGS, soymeal and canola meal were collected from the Alberta Pulse Growers Association. Price of canola oil was used from the Canola Council of Canada. Prices of other minerals and feed supplements were from Masterfeeds - retail bulk price list of July 22, 2013. Nutritional values and prices of feed ingredients are presented in Table 1.

Table 1. Nutritional values and prices of major feed ingredients

Feed	ME (kcal)	C.P (%)	Ca (%)	P (%)	Methionine (%)	Lysine (%)	Threonine (%)	Methionine +Cystine (%)	Isoleucine (%)	Tryptop han (%)	$/tonne
Wheat	3150	13	0.05	0.2	0.2	0.49	0.42	0.41	0.3	0.21	258
Barley	2780	11.5	0.1	0.2	0.21	0.31	0.4	0.42	0.5	0.19	240
Soymeal	2550	48	0.2	0.37	0.72	3.22	1.96	1.51	2.6	0.71	569
Canola meal	2000	37.5	0.65	0.45	0.69	2.21	1.72	1.3	1.4	0.5	338
Corn	3330	8.5	0.01	0.13	0.2	0.2	0.41	0.31	0.29	0.1	290
Corn DDGS	2770	36.5	0.07	0.77	0.5	0.73	0.96	1.04	0.96	0.2	305
Peas	2550	23.5	0.1	0.3	0.3	1.6	0.9	0.5	1.1	0.23	287
Minimum requirement	2800	16	4.6	0.33	0.34	0.73	0.55	0.6	0.53	0.15	

Similar to the finding the minimum cost target, the impact target is solved using the following LP model:

$$\min I = \sum_{i=1}^{I} \pi_i X_i \qquad 4$$

Subject to:

$$\sum_{i=1}^{I} a_{ij} X_i \geq b_{ij} \;\forall\; j = 1,2,\ldots, J-1 \qquad 5$$

$$\sum_{i=1}^{I} a_{ij} X_i \leq b_j \qquad 6$$

The objective function in equation 4 describes the summation of the carbon footprint of the i feed ingredients (π_i) multiplied by its amount (X_i) used in the optimal feed formulation. Equation 5 and 6 are specified by similar procedures to equation 2 and 3.

When the target values (C* and I*) are solved, a multiple objective programming model is solved using the following MINIMAX formulation:

$$Min\ \lambda \tag{7}$$

Subject to:

$$\sum_{i=1}^{I} a_{ij} X_i \geq b_{ij}\ \forall\ j = 1,2,...,J-1 \tag{8}$$

$$\sum_{i=1}^{I} a_{ij} X_i \leq b_j \tag{9}$$

$$\sum_{i=1}^{I} \pi_i X_i = C \tag{10}$$

$$\sum_{i=1}^{I} \pi_i X_i = I \tag{11}$$

$$w_C (C - C^*)/C^* \leq \lambda \tag{12}$$

$$w_I (I - I^*)/I^* \leq \lambda \tag{13}$$

The objective function of the multiple objective model minimizes λ ,which is the weighted percentage deviations from the target values for each of the two objectives. The constraints specified by equation 12 and 13 measure percentage deviations from the target values when the weights for the goals $(W_C \text{ and } W_I)$ are equal to one. However, the model is flexible to adjust the weights that reflect a decision maker's priority over each objective.

3. Results

3.1. Carbon footprint of feed ingredients

Results of global warming potential (GWP 100a) ranged from 0.012 to 1.799 kg CO_2e/kg of feed ingredient (Table 2). Canola oil had the highest GWP while limestone had the lowest GWP. Among major feed crops, corn DDGS had the highest GWP, followed by wheat and corn which had greater GWP than barley, canola meal, peas and soymeal. For major protein feed ingredient, GWP of soymeal was higher than that of canola meal. Fertilizer production and its emissions from field application was a major contributor to GWP of feed crops. Transportation was mainly responsible for GWP of imported feed grains such as corn, corn DDGS and soymeal which accounted for 41, 33 and 19 % of total impact of GWP.

3.2. Least cost feed formulation

Least cost feed formulation had the lowest cost of $282/tonne and 562 kg CO_2e/tonne. The feed formulation contained wheat (18%), barley (25%), canola meal (8.7%), peas (19.7%), corn DDGS (17%), limestone (11%) and others[1] (1.3%). As energy and protein are essential for layer feed, unit cost/crude protein (%) and energy (1000 kcal) are calculated and presented in Figure 1. Wheat, barley and corn had the lower level of cost per unit energy while soymeal, canola meal, peas and corn DDGS had the lower cost per unit protein. Corn had the highest cost per protein content and soymeal had the highest cost per energy content. Both corn and soymeal had a huge gap of trade-off between energy and protein content. In terms of cost minimization, corn DDGS was more competitive than corn.

[1] Others included canola oil, dicalcium phosphate, salt and vitamin premix.

Table 2. Carbon footprint of feed ingredients (kg CO_2e/tonne)

Feed	kg CO_2e/tonne
Wheat	662
Barley	425
Soymeal	541
Canola meal	406
Corn	656
Corn DDGS	1153
Peas	435
Limestone	12
Canola oil	1799
Dicalcium phosphate	970
Salt	226
Vitamin premix	680

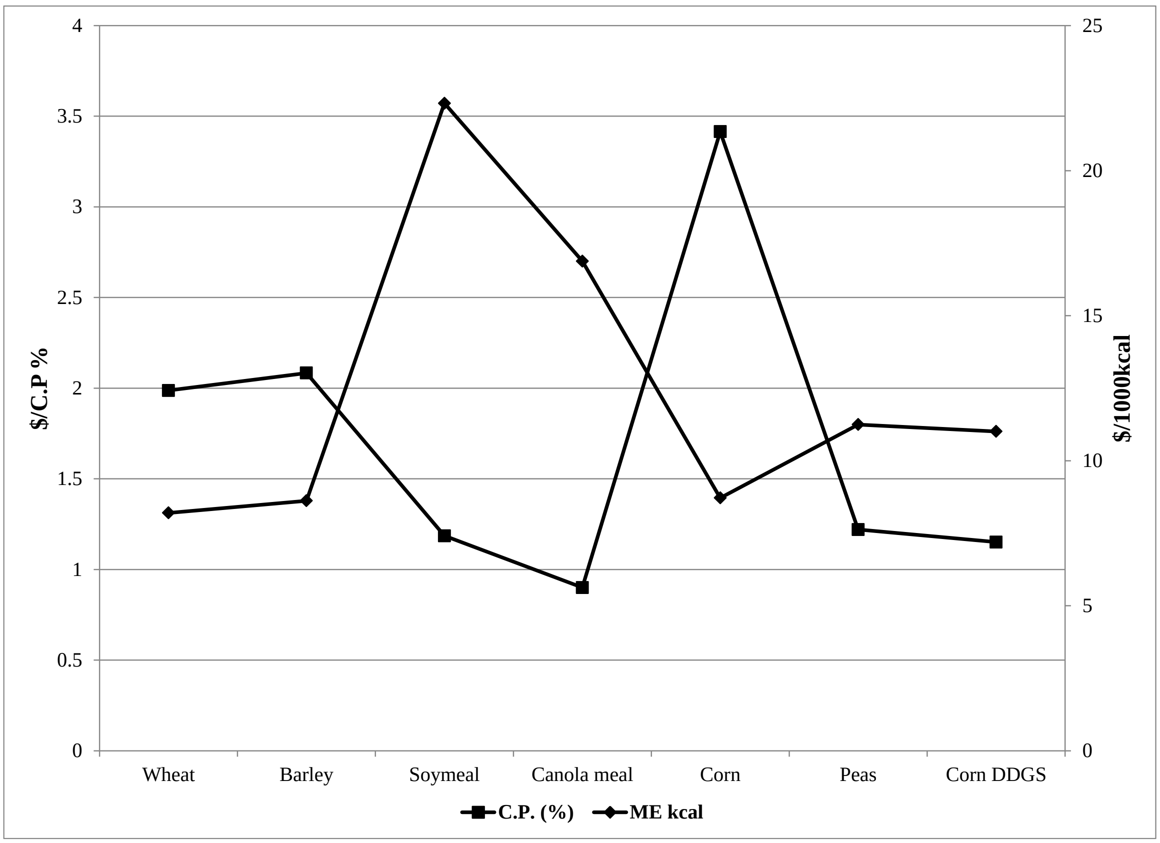

Figure 1. Unit cost per crude protein (%) and energy (1000 kcal) of major feed ingredients

3.3. Low-impact feed formulation

Low-impact feed formulation had the minimum level of 403 CO_2e/tonne and feed cost of $296/tonne. Low-impact feed reduced the carbon coefficient by 159 kg CO_2e/tonne (28%) at an increase cost of $13/tonne (4.5%). The feed formulation contained barley (59%), soymeal (9.5%), canola meal (8.5%), peas (12%), limestone (11%) and others (1.3%). Carbon footprint per crude protein (%) and energy (1000 kcal) are presented in Figure 2. Soymeal and canola meal were more competitive than peas and corn DDGS for crude protein. Wheat, corn and corn DDGS had a higher carbon footprint per unit crude protein and energy content. Barley, soymeal, canola meal and peas had the lower level of carbon footprint per both nutritional values.

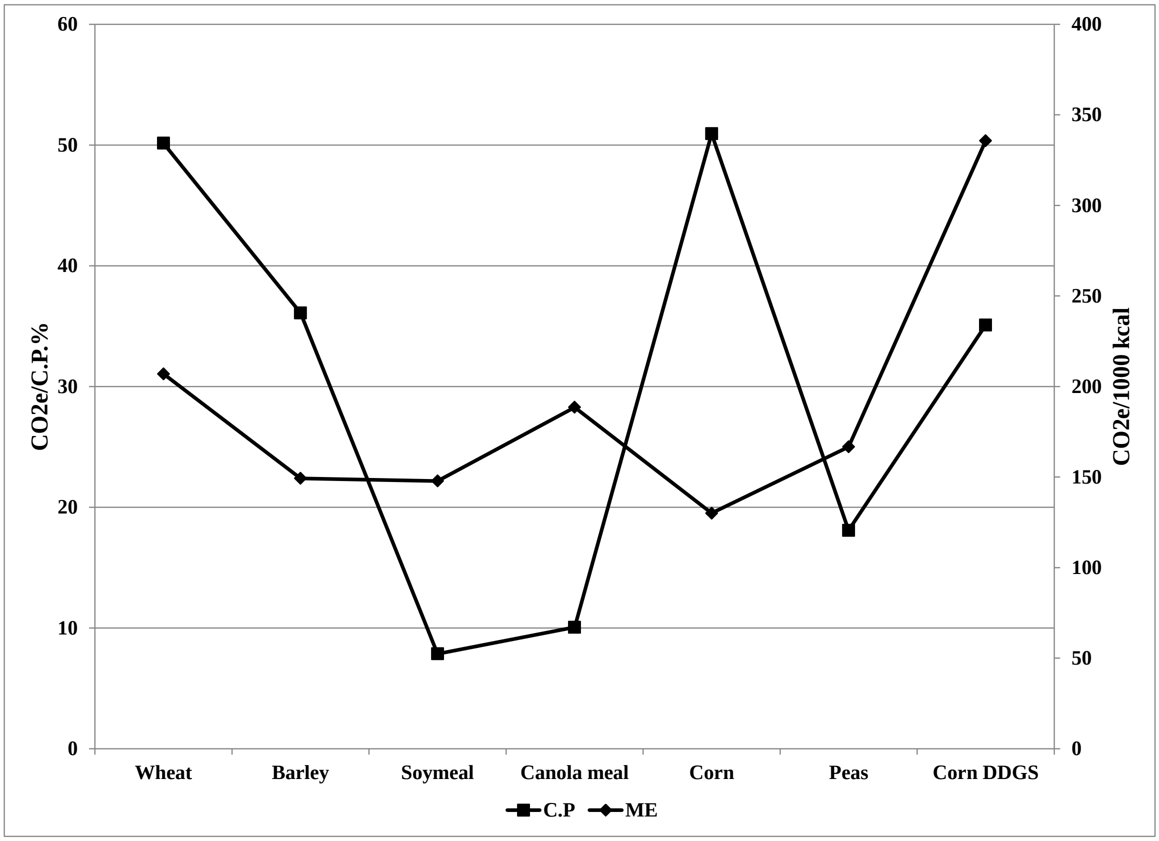

Figure 2. Carbon footprint per unit of crude protein (%) and metabolizable energy (1000 kcal) for seven major feed ingredients

3.4. Multiple objective solutions

The lowest cost ($282/tonne) from the least cost feed formulation and the minimum level of carbon footprint (403 CO_2e/tonne) from the low-impact feed formulation were used as the target levels of the two objectives – least-cost and low-impact. Multiple objective programming was solved under three different weighting scenarios for the two targets – equally weighted scenario (W_{LC} = W_{LI}= 1), cost heavily weighted scenario (W_{LC} = 2, W_{LI}= 1) and carbon footprint heavily weighted scenario (W_{LC} = 1, W_{LI}= 2).

The results of least cost, low-impact and three multiple objective solutions are presented in Figures 3 and 4. For the solution of equally weighted scenario (W_cW_i), the cost of feed formulation was $291/tonne, an increase over the least cost formulation of $9/tonne or 3% and, the carbon footprint was 411 kg CO_2e/tonne with a marginal decrease of 151kg CO_2e/tonne or 27% compared to the least cost formulation (LC). For the solution of the carbon footprint heavily weighted scenario ($1W_c2W_i$), the cost and carbon footprint of feed formulation were $296/tonne and 403 kg CO_2e/tonne that increased the cost by $14/tonne or 5% and decreased the footprint by 159 kg CO_2e/tonne or 28% compared to the least cost formulation. The results were almost the same as those of the low-impact (LI) feed formulation ($295/tonne and 403 kg CO_2e/tonne). Compared to the results of the equally weighted solution (a 3% cost increase and a 27% CO_2e reduction), the low-impact and the carbon footprint heavily weighted scenarios cost more (a 5% cost increase) but reduced relatively similar amount of the carbon footprint (a 28% CO_2e reduction). Therefore, both solutions are not optimal in terms of the marginal changes in cost and carbon footprint.

For the solution of the cost heavily weighted scenario ($2W_c1W_i$), the cost of feed formulation increased by only $1/tonne or 0.35% compared to the least cost formulation but the carbon footprint decreased by 105 kg CO_2e/tonne or 19%. The solution of the cost heavily weighted scenario was more feasible than that of the equally weighted scenario in terms of the marginal changes in cost and carbon footprint. Compared to the least cost formulation, a major change in these formulations was the substitution of wheat and corn DDGS by barley, peas and soymeal. The results indicated that wheat and corn DDGS had relatively higher carbon footprints with respect to crude protein and energy content in the diet than did other feed ingredients.

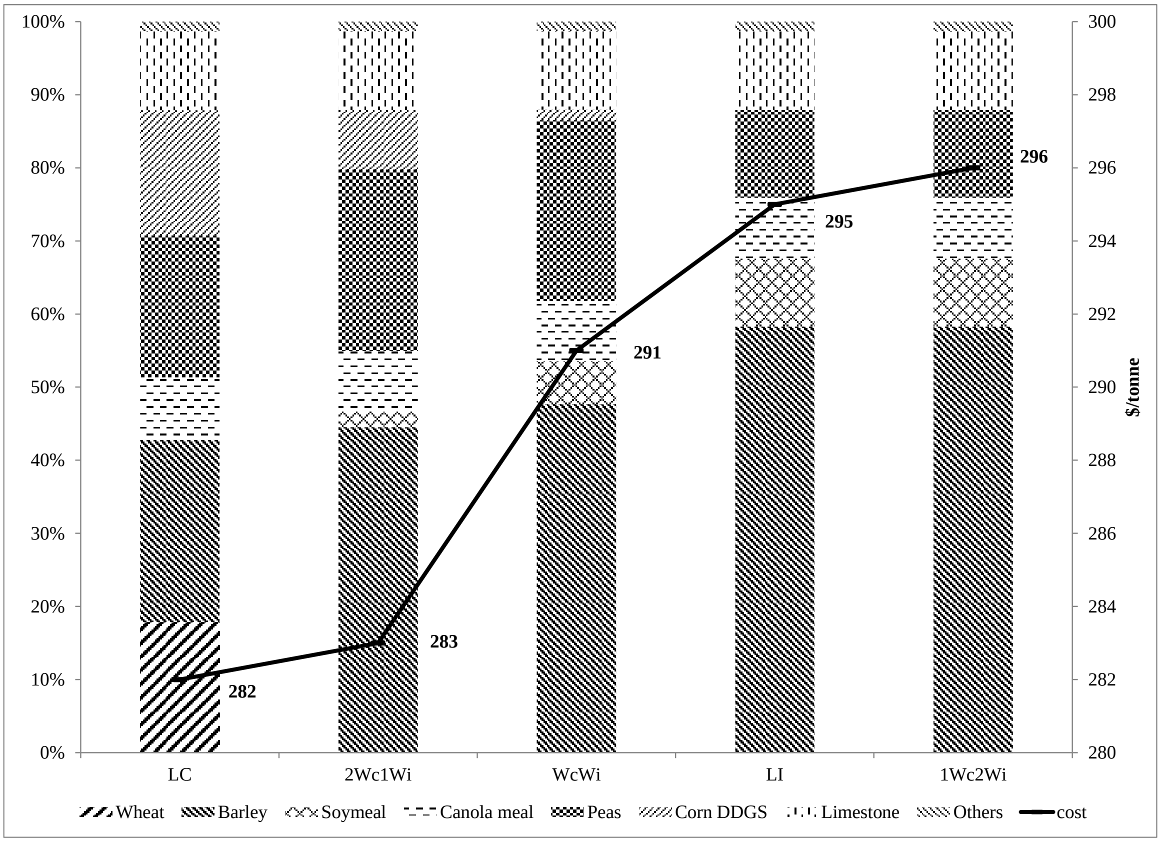

Figure 3 Optimal feed formulations and costs for five different scenarios

Note: LC, LI, W_cW_i, $2W_c1W_i$ and $1W_c2W_i$ stand for least cost scenario, low-impact scenario, equally weighted multiple objective scenario, cost heavily weighted multiple objective scenario and carbon footprint heavily weighted multiple objective scenario respectively.

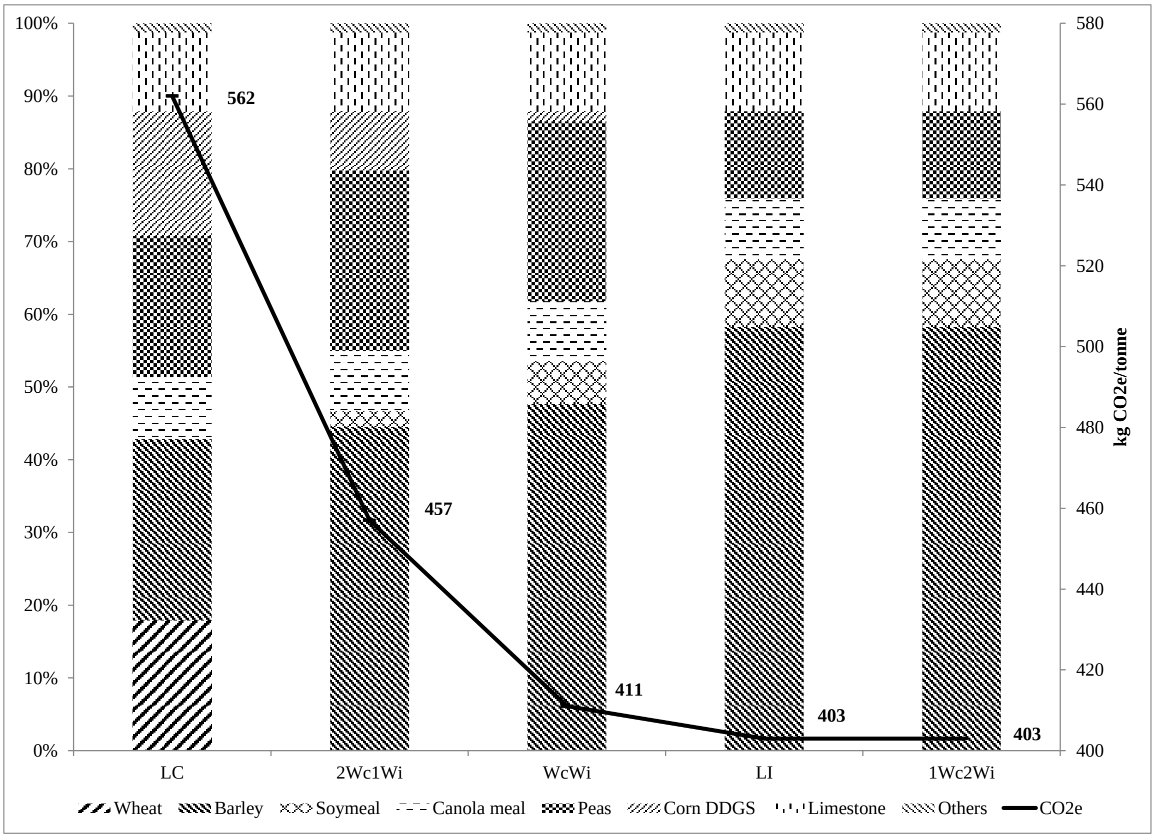

Figure 4 Optimal feed formulations and carbon footprints for five different scenarios

Note: LC, LI, W_cW_i, $2W_c1W_i$ and $1W_c2W_i$ stand for least cost scenario, low-impact scenario, equally weighted multiple objective scenario, cost heavily weighted multiple objective scenario and carbon footprint heavily weighted multiple objective scenario respectively.

4. Discussion

As feed production contributes to a large environmental impact of livestock production, feed has become a targeted environmental hotspot of attention in livestock production systems. The environmental impact of feed production could be reduced by supplementation with synthetic amino acids and inclusion of locally grown and low impact feed ingredients in feed formulations (Eriksson et al. 2005; Mosnier et al. 2011; Nguyen et al. 2012). Mosnier et al. (2011) found that the incorporation of synthetic amino acids in pig and broiler feeds could reduce the use of high impact Brazilian soymeal, resulting in a lower carbon footprint of feed formulations. Eriksson et al. (2005) investigated the impact of feed choice of pig production using three alternative scenarios of protein supply. The results indicated that feed formulation based on peas and rapeseed meal supplemented with synthetic amino acid was environmentally preferable because of the exclusion of high impact soymeal from the feed. Nguyen et al. (2012) confirmed that carbon footprint and eutrophication potential of poultry feeds could be reduced by the substitution of soymeal and cereals by rapeseed meal, grain legumes and co-products (wheat bran, gluten) with about a 2-8% increased cost.

Similar to the study of Nguyen et al. (2012), the results of this study suggest that carbon footprint of feed formulation could be reduced with a modest increase in the feed cost. Each feed ingredient has the strength and weakness in terms of cost and carbon footprint with respect to protein and energy content. Corn DDGS is more competitive than corn because corn DDGS contains a relatively high level of energy and protein compared to corn which contains more energy and less protein. Wheat is also a competitive feed ingredient because the unit cost per protein and energy is similar to that of barley. However, the low-impact formulation does not include wheat and corn DDGS because both feed crops have a higher carbon footprint. Barley, peas and canola meal have become major feed ingredients in terms of both cost minimization and reduction of carbon footprint. However, the inclusion of large proportion of barley in feed formulation requires adding non-starch polysaccharide (NSP) enzymes to layer feeds to enhance energy digestibility (Geraert and Dalibard 2003). Peas and canola have limitations of inclusion in formulations so that the amount of peas cannot exceed more than 30% and the amount of canola meal cannot exceed more than 10% (Hickling 2003; Newkirk 2009). Therefore, it is important to consult with poultry nutritionists about any suggested changes in feed formulations so that nutritional requirements and ingredient limitations are not compromised.

5. Conclusion

Carbon footprints of feed formulations could be reduced using low impact feed ingredients while satisfying nutritional requirement of laying hens. The feed cost will be higher in order to reduce carbon footprint of feed formulation. However, it is possible to find a lower carbon footprint of formulated feed at a reasonable cost. Barley, peas and canola meal play a major role to reduce the carbon footprint of feed formulation. Multi-criteria decision analysis could be used as a tool to achieve the best possible combination of feed ingredients at the lowest possible cost using multiple objective programming. This tool will help Alberta egg producers develop eco-efficient feed formulations that demonstrates industry's commitment to environmental sustainability.

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Implications of increasing demand for freshwater use from the water footprint of irrigated potato production in Alberta

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ABSTRACT

Rising food demand, rapid urbanization, industrial development and climate change all lead to pressure on freshwater resources. Agriculture is one of the largest users of global freshwater resources, accounting for about 70% of freshwater withdrawals for irrigation. Irrigation accounts for 84% of the total surface water use in the South Saskatchewan River Basin (SSRB) in Alberta. A projection of 53% increase in freshwater use in the SSRB by 2030 could lead to additional pressure on freshwater resources in the region. This study assesses the impact of current and future freshwater use on regional water deprivation impact potential. The result suggests that the water stress index (WSI) and the blue water footprints for a future freshwater use scenario were three times higher than those for the current freshwater use scenario. A combination of increasing irrigation efficiency and improving crop water use efficiency will be an effective way to mitigate pressures of water stress.

Keywords: Water footprint, Water deprivation potential, Freshwater, Irrigated potato, Alberta

1. Introduction

Freshwater use has become a major social and environmental concern in the last two decades. This is due to rising food demand, rapid urbanization, industrial development and climate change which are significantly increasing pressure on freshwater resources (Ridoutt and Pfister 2010; Gheewala et al. 2013). Agriculture is one of the largest users of freshwater resources globally, accounting for about 70% of freshwater withdrawals for irrigation. Alberta has approximately 70% of the irrigated land in Canada. Alberta sources all of its irrigation water from rivers and reservoirs which accounts for 84% of the total freshwater use in the South Saskatchewan River Basin (SSRB) in Alberta (AMEC 2009). Irrigation water use is competing with other demands for freshwater such as urban and industrial consumption.

Global scenarios for future water demand and supply have identified major drivers of change in water demand in order to understand possible changes in the water footprint of production and consumption and to formulate alternative strategies for meeting increased demand for water (de Fraiture and Wichelns 2010; Ercin and Hoekstra 2014). The results of the scenario analyses suggested that future demand for freshwater use would increase significantly and it could exacerbate freshwater scarcity problems unless appropriate action was taken to improve irrigation efficiency and water use efficiency.

Similar to the global scenario analyses, a study of current and future water use in Alberta was conducted to establish a baseline assessment of current water use, to project future water use and to identify potential improvement strategies to meet future freshwater demand. A projection of water supply for the SSRB in Alberta forecasts that water use in the basin will increase 53% from the current 1.98 km^3 to about 3.04 km^3 by 2030 mainly due to expansion of irrigation districts (AMEC 2009). The water demand and supply study did not address a question of how future increased freshwater demand will impact the water footprint of primary agricultural production in Alberta. Therefore, this study assesses the impact of future freshwater use on the water footprint of irrigated potato production in Alberta and identifies potential mitigation strategies to alleviate future water deprivation potential. Potato production is chosen for the study because it is a high value crop, one of the major irrigated speciality crops in southern Alberta and processed potatoes are primarily dependent on irrigation water.

2. Methods

2.1. Water stress index

The water stress index (WSI) developed by Pfister et al. (2009) was used to calculate regional water stress characterization factor relevant to the SSRB in Alberta where most of the irrigated potato production exits. WSI is adjusted to a logistic function ranging from 0.01 to 1 (0.01=low water scarcity, 1= high stress) using a ratio of total annual freshwater withdrawal to hydrological availability (WTA) with modifications to account for monthly and annual variability of precipitation and corrections to account for watersheds with strongly regulated flows (Ridoutt and Pfister 2010).

AMEC (2009) assessed current and future water supply and demand in the SSRB with the Water Resource Management Model (WRMM). The model considered water supplies, natural water flows, precipitation, configuration of streams, canals and water management infrastructure, consumptive and in stream demands, water license priorities, water management policies and operating plans. The SSRB is comprised of four sub-basins: Red Deer, Bow, Oldman and South Saskatchewan. For future water use in 2030, it was assumed that there would be a 32% expansion of irrigation district areas in the Bow sub-basin and a 19% expansion in the Oldman sub-basin. Current water availability, current water use and future water use of the SSRB were taken from AMEC's Water Resource Management Model to calculate the withdrawals to hydrological availability (WTA).

The calculated WTA was adjusted by variation factors of monthly and annual precipitation of four sub-basins in the SSRB. Monthly and annual precipitation normals (1961-1990) were obtained from the AgroClimatic Information Service (ACIS) of Alberta Agriculture and Rural Development (ARD). Coefficients of variation of monthly and annual precipitation for each water sub-basin were calculated using 1600 township weather data[1]. The variation factor of the SSRB was calculated using a weighted average of variation factors and the mean annual precipitation of four sub-basins.

The water stress indices of the whole sub-basins in the SSRB were calculated for current water use and future water use scenarios by using the WTA for both scenarios and the variation factor. The WSI was used as a midpoint characterization factor to measure the water deprivation impact potential. As this study focuses on freshwater use and its impact on regional freshwater deprivation potential, the water deprivation potential is calculated by multiplying the blue water footprint with the WSI for the current and future use scenarios.

2.2. Water footprint

In terms of virtual water, water consumption is divided into three categories: green, blue and grey (Hoekstra et al. 2011). The green water consumption describes the evapotranspiration of rainwater during plant growth, which is especially relevant for agricultural products. Blue water consumption describes the volume of ground and surface water that evaporates during production, especially important for irrigated agriculture systems (Ridoutt and Pfister 2010). Grey water consumption describes the total amount of water required to dilute the used water until it reaches commonly agreed quality standards. Grey water consumption is excluded from this study because other LCA studies measure the impact of grey water as eutrophication and acidification potential in life cycle impact assessment (LCIA).

Crop water requirements of potato production for six locations from three irrigation districts (2011) were obtained from the Alberta Irrigation Management Model (AIMM, ARD 2014). The AIMM calculates daily evapotranspiration values over a growing season (May 15 to September 30) using daily climate data from the climate information network stations and crop coefficients and reference evapotranspiration of Penman-Monteith equations developed by Allen et al. (1998). Green and blue water footprints were calculated using the water footprint assessment method developed by Hoekstra et al. (2011). Irrigated potato yield for 2011 was obtained from Agriculture Financial Service Corporation (AFSC, 2012).

The US potato production data were adapted from ecoinvent database to estimate indirect blue water consumption for crop inputs and field operations (Nemecek and Kagi 2007). Data on field operations and

[1] Township weather data were calculated using a mathematical data interpolation method that weighted up to the 8 nearest station observations (ACIS 2013).

transportation was used from the US potato production because cultivation methods were similar across the North American commercial potato production system. Fertilizer production was modified using Alberta/Canada electricity grid mix. Fertilizer application rates were used from fertilizer recommendations for irrigated potato by Alberta Agriculture and Rural Development. Pesticide use was calculated from the 2011 Potato Crop Weed and Pest Control Guide, Prince Edward Island Department of Agriculture. Indirect blue water consumption of potato production was calculated by Impact 2002+ version 2.2 method in SimaPro 7.3.3 (Humbert et al. 2012; Goedkoop et al. 2010).

2.3. Sensitivity analysis

Sensitivity analyses of the future blue water footprint were conducted to identify potential mitigation strategies for alleviation of freshwater deprivation potential. Irrigation water conveyance efficiency, application efficiency and water use efficiency were also included in the analysis. The first sensitivity analysis included a combination of 4.3% increase in conveyance efficiency and 15% increase in application efficiency[2]. It is assumed that irrigation efficiency could be increased by improving irrigation infrastructure and adopting more efficient application equipment. The second and third sensitivity analyses included a 10% and 20% increase in potato yield (increased water use efficiency). It is assumed that potato yields could be increased by varietal and agronomic improvements.

3. Results

Water stress indices (WSI) of South Saskatchewan River Basin were 0.2078 for the current water use scenario and 0.5981 for the future water use scenario in 2030. An increase of 53% in future freshwater use could change from the current very mild water deprivation potential to a moderate severe water deprivation potential in the SSRB because a higher level of water withdrawal leads to a higher value of freshwater WTA, resulting in a higher value of WSI. The results of WSI for current and future freshwater use scenarios suggest that an expansion of irrigated areas in the SSRB would necessitate a determination of what is a sustainable level of irrigation withdrawals from the basin.

Irrigation water requirements of potato production ranged from 146 to 353 mm yr^{-1} depending on potato evapotranspiration (mm) and precipitation (mm) (Table 1). Grassy Lake required the lowest level of irrigation water because it had the highest level of precipitation. Conversely, Seven Persons required the highest level of irrigation water because of the lowest level of precipitation in the region. Both Grassy Lake and Seven Persons were located in the South Saskatchewan sub-basin. Total irrigation water requirement of potato production ranged from 179 to 433 mm after conveyance and application water losses were taken into account.

The total water footprint of potato production ranged from 133 to 158 L/kg (Table 2). As previously mentioned, the study focused on freshwater use and its impact on regional water deprivation potential so the water deprivation impact potential of blue water was calculated multiplying blue water with the water stress index (WSI) of the SSRB. The stress-adjusted blue water footprints are presented in Table 3. The results of the future freshwater use scenario in all regions were about three times greater than the current freshwater use scenario because the value of the WSI for future water use was about three times higher than those of current water use.

[2] Water conveyance efficiency is the ratio between the irrigation water that reaches a farm or field to that diverted from the water source. Water application efficiency is a measure of the fraction of the total volume of water delivered to the farm or field to that which is stored in the root zone to meet the crop evapotranspiration needs (Irmak et al. 2011).

Table 1. Parameters used for calculation of irrigation water requirement for irrigated potato production

Irrigation District	St. Mary	St. Mary	St. Mary	Taber	Taber	Bow River
Location	Seven Persons	Bow Island	Grassy Lake	Fincastle	Barnwell	Enchant
Potato yield (tonne/ha)	41	41	41	41	41	41
Conveyance efficiency (%)	93.7	93.7	93.7	93.7	93.7	93.7
Irrigation application efficiency (%)	80	80	80	80	80	80
Evapotranspiration (mm)	486	435	441	484	478	437
Precipitation (mm)	133	155	295	187	260	172
Irrigation water requirement (mm)	353	283	146	297	218	265
Conveyance loss (mm)	9	7	4	8	6	7
Irrigation application loss (mm)	71	56	29	59	44	53
Total irrigation water requirement (mm)	433	343	179	364	267	325

Table 2. Water footprint of irrigated potato (per hectare and per kg of potato)

Irrigation District	St. Mary	St. Mary	St. Mary	Taber	Taber	Bow River
Location	Seven Persons	Bow Island	Grassy Lake	Fincastle	Barnwell	Enchant
Green WF (m^3/ha)	1330	1550	2950	1870	2600	1720
Direct blue WF (m^3/ha)	4458	3536	1844	3751	2753	3347
Total WF (m^3/ha)	5788	5086	4794	5621	5353	5067
Green WF (L/kg)	33	38	73	46	64	42
Direct blue WF (L/kg)	110	87	45	92	68	82
Indirect blue WF (L/kg)	15	15	15	15	15	15
Total WF (L/kg)	158	140	133	153	147	139

Table 3. Water Stress Index-weighted blue water footprint of irrigated potato (L/kg) for two different water use scenarios

Irrigation District	St. Mary	St. Mary	St. Mary	Taber	Taber	Bow River
Location	Seven Persons	Bow Island	Grassy Lake	Fincastle	Barnwell	Enchant
WSI=0.2078 current water use	26	21	13	22	17	20
WSI=0.5981 future water use	75	61	36	64	50	58

Results of the sensitivity analyses are presented in Table 4. The results confirmed that an increase in irrigation efficiency and water use efficiency could alleviate future freshwater deprivation potential of potato production. Increased irrigation efficiency reduced about 11-13% in the blue water footprint. A 10% potato yield increase reduced the blue water footprint by 6-8%. And a 20% potato yield increase reduced about 22-25% of the blue water footprint.

Table 4. Sensitivity analyses for irrigation efficiency and potato water use efficiency (blue water footprint L/kg)

Irrigation District	St. Mary	St. Mary	St. Mary	Taber	Taber	Bow River
Location	Seven Persons	Bow Island	Grassy Lake	Fincastle	Barnwell	Enchant
Baseline	75	61	36	64	50	58
Increased irrigation efficiency	65	53	32	56	43	51
10% yield increase	69	56	34	59	46	54
20% yield increase	56	46	28	48	38	44

4. Discussion

An increase in irrigation freshwater use could lead to a moderate-severe level of regional freshwater scarcity. A projection of a 53% increase in freshwater demand by 2030 is not sustainable in terms of regional freshwater withdrawals to hydrological availability. However, sensitivity analyses indicate that increased irrigation efficiency and water use efficiency could alleviate the severity of regional freshwater scarcity.

In order to maintain sustainable consumption of freshwater resources in the region, it is necessary to develop strategies for sustainable freshwater resource management including improvement in irrigation efficiency, increased water use efficiency and the development of new drought resistant and short growing season varieties (Ridoutt and Pfister 2010; Gheewala et al. 2013). Although most potato growers already use efficient irrigation systems (low pressure pivot sprinkler systems) in the SSRB, there is still room for improvement in irrigation efficiency through adoption of new advanced irrigation systems such as LEPA (low energy precise application) sprinkler irrigation systems, variable rate irrigation systems (VRI) and, advanced management such as adequate irrigation scheduling.

Crop water use efficiency plays a major role in alleviating water scarcity because irrigation water requirements of crop production mainly depend on crop evapotranspiration (Gheewala et al. 2013). The development of drought resistant and high-yield crop varieties could potentially minimize the water footprint of crop production. Figure 1 illustrates that the average potato yields have increased by about 25% over the past the two decades due to varietal and agronomic improvements. Improvement in crop yield could lower the water footprint of agricultural products. An integrated management of freshwater resources combining strategies for increasing irrigation efficiency and improving crop water use efficiency will be an effective way of reducing the blue water footprint of irrigated crop production and alleviating the water deprivation potential in the region.

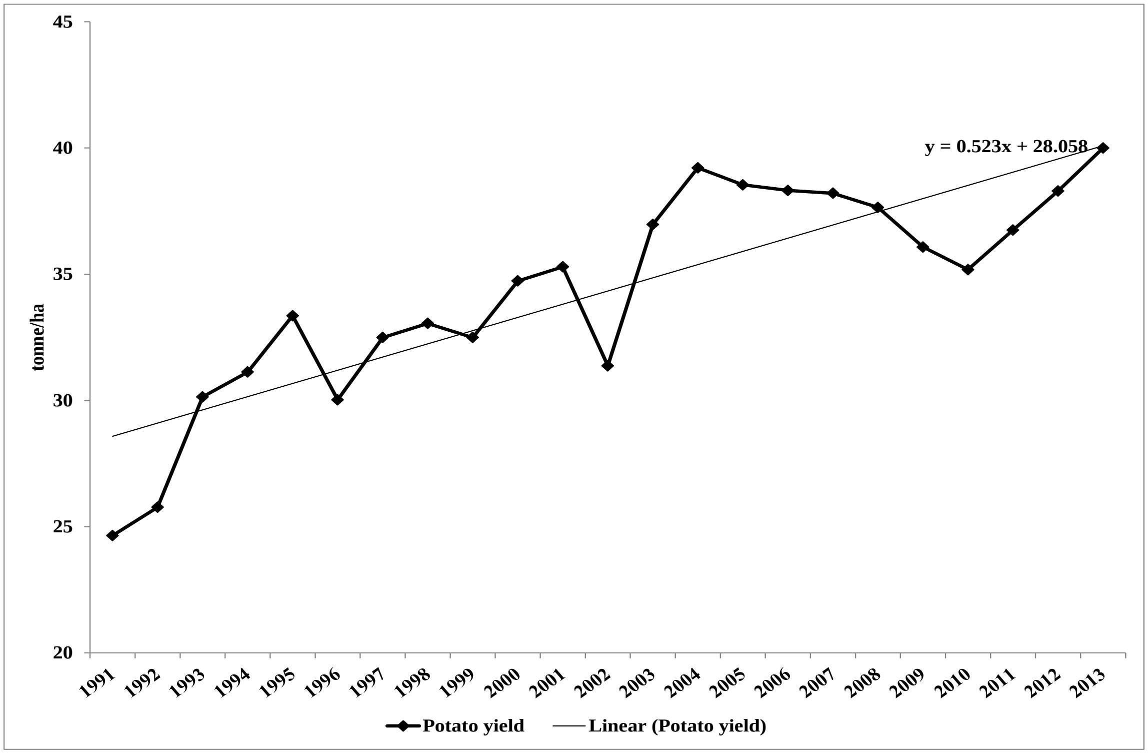

Figure 1. Average potato yield in Alberta (1991-2013)
Source: Statistics Canada, CANSIM Database (1991-2013)

5. Conclusion

Increased water use for irrigation could lead to additional pressure on freshwater resources and in a greater blue water footprint of irrigated potato production. A regional water stress index is a relevant characterization factor to measure local water scarcity in the particular region. Therefore, the WSI can be used to assess the impacts of different scenarios of water use and supply on regional water deprivation potential. A combination of increased irrigation efficiency and an improvement in crop water use efficiency will be essential to mitigate increase pressures of water stress.

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A method to handle emissions related to manure production in LCAs with an example from a steer production system

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ABSTRACT
Different methods have been used in LCAs to handle emissions related to livestock manure. Usually, the emissions from use of manure are attributed to the crop production. In this study, an alternative approach is suggested where manure is considered as a co-product from the livestock production. With this approach, the livestock production system 'pays' all environmental costs related to emissions from manure. However, the livestock system also gets credit for the fertilizer value of the manure corresponding to the amount of artificial fertiliser nitrogen that the farmer would otherwise have applied. As an example of using this approach, carbon footprint was calculated from four Danish steer production systems that only differ in housing system/type of manure produced. The steer production systems were: '100% outdoor/grazing', '100% indoor/deep litter', '100% indoor/slurry', or a mix as in the 'real system' from Danish herds.

Keywords: Carbon footprint, LCA methodology, Livestock manure production, Soil carbon changes,

1. Introduction

Different methods have been used in LCAs to handle emissions related to livestock manure, where the emissions from manure have been attributed to either crop production or livestock production. Usually, the emissions from the use of manure are attributed to crop production (van Zeijts et al., 1999). However, Dalgaard and Halberg (2007) suggested that the environmental burden of using manure should be considering as a co-product from the livestock production. This means that the livestock production system 'pays' all environmental costs related to emissions from manure. However, the livestock system also gets credit for the fertilizer value of the manure. This is also reflected in a new guideline from EU on methods for calculating the life cycle environmental performance of products (EU, 2013). The EU guideline suggests that when manure nitrogen is applied to agricultural land and directly substitutes an equivalent amount of the specific fertiliser nitrogen that the farmer would otherwise have applied, the animal husbandry system from which the manure is derived should be credited for the displaced fertiliser production (taking into account differences in transportation, handling, and emissions) (EU, 2013). The substitution rate of 'kg fertilizer N' per 'kg N in manure ex animal' is assumed to follow the Danish legislation (Anonymous, 2010).

The aim of the present study was to illustrate how an alternative approach, where manure is seen as a co-product from the livestock production, can be used to calculate environmental costs related to emissions from manure. This was illustrated by an example, where carbon footprint was calculated from four Danish steer production systems that only differ in housing system/type of manure produced.

2. Methods

To illustrate the suggested approach for including emissions related to manure production, carbon footprint was calculated from four Danish steer production systems that only differ in housing system and type of manure produced. The steer production systems were: 'a real system (1)' as found in private herds '100% outdoor/ grazing (2)', '100% indoor/deep litter (3)', and '100% indoor/slurry (4)'. In the 'real system' (data based on Nielsen, 2003), the steers were grazing for 143 days per year at semi-natural pasture resulting in a daily gain of 550 g. During winter, the youngest steers were housed at deep litter and afterwards in a slurry-based system and the feeding was based on grass clover silage and limited amount of concentrate, resulting in a growth rate of 900 g/d. For the last 45 days the steers were fed more intensively with more concentrate and were gaining 1,100 g/d before slaughtering at 25.4 months of age.

In the system '100% outdoor/ grazing', all manure was deposited at pasture. The system was similar to the 'real system', except that it was assumed that the steers were outdoor continuously, though feeding was similar to that in the 'real system'. In the '100% indoor systems', the steers were housed indoor and produced manure was either 'deep litter' or 'slurry'. Intake of fresh pasture was replaced by the same amount of energy from grass clover silage. Table 1 present the most important input and output from the steer production system. Smaller changes in the steer production exist due to the different housing/manure systems. These are mentioned in the footnotes of Table 1.

According to the new approach, manure is considered a co-product from the livestock system, in this case the steer production system. A process was defined for each of the three types of housing/manure production: 'slurry', 'deep litter', 'manure deposit at pasture'. A carbon footprint was calculated for each of the three types of housing/manure with a functional unit (FU) of '100 kg manure N ex animal' according to the method by Mogensen et al. (2014). In other words, the GHG emissions from '100 kg manure N ex animal' from different housing/manure systems was calculated. It was taken into account both the direct emissions from manure, but also that manure cause C and N sequestration, and therefore less leaching, and avoided fertilizer production. Manure N ex animal was calculated as N in animal feed minus N in gain (Nielsen & Kristensen, 2005).

The results of these three housing/manure production process were used in calculating carbon footprint of the four steer production systems, where FU was 1 kg carcass at farm gate. With the new approach, the steer production systems were treated as' landless systems' i.e. GHG contribution from feed production and livestock production was calculated independent of each other. Crops were assumed imported to the farm and grown with use of artificial fertilizer. However, this contribution was offset to a certain degree by including the fertilizer value of the co-product manure. For each of the feedstuff used (see Table 1), a carbon footprint was calculated according to the method by Mogensen et al. (2014).

GHG emissions from soil carbon changes were calculated as suggested by Petersen et al. (2013). The approach by Petersen et al. (2013) is based on a single year's addition of C (from crop residues, etc.) and the associated effect on atmospheric CO_2. Petersen et al. (2013) estimated that 10% of the C added to the soil will be sequestered in a 100-year perspective. The input of carbon to soil C was based on the input of above- and below-ground crop residues. In the present study, contribution to soil C changes was divided into the contribution from C input from crop residues and the contribution coming from C input from different types of manure.

Table 1. Input and output in the steer production system.

Per produced steer	
Input of a dairy calf (30 days), kg live weight	55
Input of feed, kg DM	
- Grass clover silage	1,690
- Straw	220
- Barley	360
- Rapeseed cake	80
- Grazing [1)]	1,440
- Milk powder	20
Total feed, kg DM	3,820
Straw for bedding, kg [2)]	1,160
Output of a steer, kg live weight	574
- Kg carcass	293

1) Grazing was used in steer system 1 and 2. In system 3 and 4 grass grazed was replaced by same amount of energy from grass clover silage
2) Amount of straw used in steer system 1, in system 2 and 3 with slatted floor and slurry produced no straw was used at all, and in system 4 2,482 kg straw per produced steer was used

Indirect land use change (iLUC) was calculated according to Audsley et al. (2009) where all use of land for crop production is assumed to increase the pressure on land use and thus causing land use change somewhere in the world. The indirect land use change causes a release of 8.5 Gt CO_2-eq per year, to which agriculture contributes 58%. This gives a contribution of 1.43 t CO_2-eq per ha when divided by the total agricultural area of 3,475 Mha (Audsley et al., 2009). In the present study, iLUC was included by multiplying land use (m^2/kg DM feed) by the iLUC factor of 143 g CO_2-eq /m^2.

3. Results and discussion

Table 2 shows the overall effect on GHG emissions of three different ways of housing/manure production; as ‘slurry’, ‘deep litter’, or as ‘manure deposit at pasture’. This overall effect of manure takes into account both the emissions from manure and the benefit from the avoided fertilized production. Contribution to GHG emission from direct and indirect emissions of N_2O and NH_3 from manure from housing, storage and application varied from 1,171 kg CO_2-eq per ‘100 kg N ex animal in a slurry-based system’ to 1,569 kg CO_2-eq per ’100 kg N ex animal in a deep litter-based system’.

Table 2. Greenhouse gas (GHG) emission from three different housing/manure systems, FU = 100 kg N ex-animal (Mod. after Mogensen et al., 2014)

Manure system [1]	**Deposit at pasture**	**Slurry**	**Deep litter**
Housing System	Outdoor	Indoor	Indoor
Emissions from manure:			
N_2O-N direct, kg [2]			
-housing	0	0.2	1.0
-storage	0	0.5	0.5
-application	2.0	1.0	1.0
NH_3-N, kg [3]			
-housing	0	8.0	15.0
-storage	0	2.2	25.0
-application	7.0	12.0	6.0
N_2O-N indirect, kg [2]			
-from NH_3-N	0.07	0.22	0.46
-from leaching [4]	0.68	0.58	0.39
Total GHG from emissions, kg CO_2-eq	**1,288**	**1,171**	**1,569**
C sequestration from manure			
N input to soil after losses, kg N [5]	90	75	58
Related C input to soil, kg C [6]	939	783	1,581
Soil C remaining in soil, kg soil C [7]	94	78	158
Total GHG from C sequestration, kg CO_2-eq [8]	**-344**	**-287**	**-579**
N from manure stored in soil and reduced leaching [9]			
N stored in soil, kg N	9.4	7.8	15.8
Saved indirect N_2O emissions, kg N_2O-N	0.07	0.06	0.12
Total GHG from avoided leaching, kg CO_2-eq	**-21**	**-17**	**-55**
Total GHG emissions from manure, kg CO_2-eq	**923**	**867**	**935**
Avoided fertilizer production:			
Fertilizer value of manure			
N, kg [10]	70	70	45
P, kg [10]	14	14	20
K, kg [10]	91	91	137
GHG from avoided fertilizer prod., kg CO_2-eq			
- N [11]	-298	-298	-191
- P [12]	-67	-67	-93
- K [13]	-54	-54	-82
GHG from avoided fertilizer prod., kg CO_2-eq	**-418**	**-418**	**-366**
Avoided emission from fertilizer			
N_2O-N_{direct}, kg from spreading [2]	0.7	0.7	0.45
NH_3-N, kg from spreading [3]	1.54	1.54	0.99
N_2O-$N_{indirect}$, kg from NH3 and leaching	0.53	0.53	0.34
GHG from avoided fertilizer emission, kg CO_2-eq	**-574**	**-574**	**-370**
Total GHG from avoided fertilizer	**-992**	**-992**	**-736**
Total GHG from 100 kg N in manure (ex animal) , kg CO_2-eq [14]	**-69**	**-125**	**199**

1) CF from import of straw is not included in this calculation
2) Calculated according to IPCC, 2006
3) Calculated according to Mikkelsen et al., 2006
4) Leaching (NO_3-N) calculated as input minus other emission
5) Input to soil is the ‘100 kg N ex animal’ minus all losses.

6) In the deep litter system there is an extra N input from N content in straw. C:N in manure deposited at pasture and in slurry 8:1 (Wesnæs et al., 2009) and C:N in deep litter of 21:1 (Osda et al., 2001) both multiplied by a factor of 1.3 (Petersen, B pers comm., 2013)
7) According to the model by Petersen et al. (2013)
8) From C to CO_2 by factor multiplication 44/12
9) Per 10 kg C stored in soil, 1 kg N is stored in soil (Sundberg et al., 1999)
10) Anonymous, 2010
11) CF of N in fertilizer: 4,25 kg CO_2/kg N (Elsgaard, 2010)
12) CF of P in fertilizer: 4,63 kg CO_2/kg P (Ecoinvent, 2010)
13) CF of K in fertilizer: 0,596 kg CO_2/kg K (Ecoinvent, 2010)
14) Taking into account both the emissions from use of manure and the saved fertilized production

Table 3. Greenhouse gas (GHG) emission of animal feed, CO_2 g/kg DM feed (modified after Mogensen et al., 2014)

	Barley	**Barley Straw**	**Rape seed cake**	**Grass clover silage**	**Grass clover grazed**
Contribution to CF					
- Growing	484	49	390	404	453
- Processing	11	1	28	0	0
- Transport	18	18	75	0	0
Total CF	**512**	**68**	**494**	**404**	**453**
C sequestration 1)	86	8	-44	-61	-226
LUC	328	33	182	173	202
CF including soil C and LUC	**926**	**109**	**632**	**516**	**429**

1) For grazed crops contribution from C input from manure deposited is included in soil C changed.

Carbon footprint per kg carcass weight is shown in Table 4 for the four Danish steer production systems that only differ in housing/manure system. GHG from manure was calculated based on the values (Table 2) per ‘100 kg N ex animal’ corrected to actual amount of N ex animal in each steer production system and taking into account the distribution between the different housing/manure systems. Before including contribution from soil C and iLUC, CF per kg carcass was estimated to 16.6 kg CO_2/kg carcass in the ‘real steer system’, where the steers are grazing during summer and housed indoor during winter, i.e. manure system include a mixture of manure deposited at pasture, as slurry and as deep litter. Similar level of CF per kg carcass was estimated for the ‘100% outdoor system’. Here the steers are outdoor all year, but fed the same way, i.e. same contribution to CF from feed production. Lower emissions from manure from stable and storage in system 2, and lower CH_4 from manure deposited at pasture compared with as slurry was counterbalanced by less credit from substitution of fertilizer. CF per kg carcass was higher in steer system 3 with manure handled as deep litter compared with system 1 and 2. This was mainly due to higher contribution from enteric fermentation as the digestibility of grass clover silage is lower than that of fresh grass clover, which increases methane emission. Beside that there was a higher GHG contribution from production of straw for bedding. Before taking into account contribution from soil C and iLUC, steer system 4 has the lowest CF per kg carcass of all systems, even though GHG from enteric fermentation was at the same high level as in system 3, but total GHG from manure handling was lowest in the slurry-based system.

However, if GHG contribution from changes in soil carbon due to input to soil C from crop residues (‘soil C from feed’) and C from manure (‘soil C from manure’), steer system 1 ‘the real steer system’ has the lowest CF/kg carcass weight. Steer systems 4 and 2 have almost similar CF and system 3; the ‘deep litter system’ has the highest CF per kg meat even though manure as deep litter has a huge positive effect due to soil C sequestration. Including contribution from LUC did not change the ranking of the four steer production systems.

Table 4. Carbon footprint (CF) in four steer production systems that only differ in the way of housing/type of manure, CO_2 kg/kg carcass

Steer system	**1 The real system**	**2 100% outdoor**	**3 100% indoor Deep litter**	**4 100% indoor Slurry**
N in feed, kg N [4)]	117	117	107	107
N in gain, kg N	14	14	14	14
N ex animal, kg N	103	103	93	93
Manure system, % of N ex animal				
- deposited at pasture	**39**	**100**	**0**	**0**
- slurry	**15**	**0**	**0**	**100**
- deep litter	**45**	**0**	**100**	**0**
GHG from feed production, kg CO_2/kg carcass				
- growing, processing, transport	6.1	6.1	6.3	6.3
- soil C	-1.4	-1.4	-0.6	-0.6
Total GHG from feed, kg CO_2	4.7	4.7	5.7	5.7
GHG from manure, kg CO_2				
- Emissions	3.4	2.8	4.1	2.4
- Saved fertilizer production [2)]	-1.0	-0.7	-0.6	-1.1
- Effect on soil C	-1.3	-0.8	-2.5	-1.0
Total GHG from manure	1.1	1.3	1.0	0.3
GHG from CH4 enteric, kg CO_2/kg carcass	7.2	7.2	7.7	7.7
GHG from input of calf	0.6	0.6	0.6	0.6
Others, straw, minerals [3)]	0.3	0.1	0.6	0.1
CF, kg CO_2/kg Carcass	**16.6**	**16.6**	**18.6**	**15.9**
CF incl. soil C from feed, kg CO_2 [1)]	15.2	15.2	18.0	15.3
CF incl. soil C from feed and manure kg CO_2	13.9	14.4	15.5	14.3
CF incl. soil C and LUC	**16.4**	**16.9**	**18.0**	**16.8**

1) For grazed crops contribution from C input from manure deposited is included in soil C change of the crop ‘grazed grass’.
2) In system 2 only 50% of manure deposited at pasture is assumed utilized for grass production, for the other 50% no fertilizer value was assumed, however this manure contribute to soil C input (included in ‘GHG from manure’
3) No straw was used in system 2,
4) In system 3 and 4 intake of pasture is replaced by same energy form intake of grass clover silage, which lower N intake

5. Conclusion

The present paper has illustrated an approach, where the environmental burden of manure could be handled by considering it as a co-product from the livestock production. That means that the livestock production system ‘pays’ all environmental costs related to emissions from manure, and on the other hand that the livestock system also gets credit for the fertilizer value of the produced manure. Thereby, the calculated carbon footprint of feed crops is independent by use of manure or not.

The total GHG emissions from three types of manure when taking into account both the emissions from manure handling, the positive effect of soil C sequestration and reduced N leaching due to use of manure and the benefit from the avoided fertilized production, was investigated. Production and use of manure as either slurry or deposited at pasture generated a positive effect for the environment compared with use of fertilizer, whereas the opposite was seen for producing and use of manure in a deep litter system. It was found that the total GHG emission from using manure in term of ‘slurry’ will cause a reduction in GHG emission of 125 kg CO_2-eq per 100 kg N ex animal. Similar, the overall effect was a saved GHG emission of 69 kg CO_2-eq/100 kg N ex animal if manure was deposited at pasture. Whereas there was an overall GHG release of 199 kg CO_2-eq per 100 kg N ex animal as deep litter. There estimates are under the assumption that the manure was used in a way that it will be utilized to substitute use of fertilizer. Due to that, carbon footprint of identical steer production system that only differ in the type of housing/manure system, came out with a lower CF in a steer production systems, where the type of manure was ‘slurry’ or ‘deposited at pasture’ compared with CF of a steer production system on deep litter.

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A novel approach to assess efficiency of land use by livestock to produce human food

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ABSTRACT
The increasing demand for livestock products will intensify the claim on land. To use land efficiently, we need a method to determine the efficiency of land use to produce animal-source food. Life cycle assessment (LCA) has been used to assess the land use and shows that land required to produce, for example, 1 kg of protein is in the same range for eggs and milk. Methods used, however, do not take into account that laying hen diets contain more human-edible plant products than dairy cow diets. Furthermore, the suitability of land to grow human-edible plant products is not taken into account, as all land is counted equal. The aim of this study, therefore, was to develop a method to determine the efficiency of land use by livestock for human food production. We illustrated our novel approach for the case of egg and milk production systems in the Netherlands.

Keywords: animal production, land use efficiency, egg, milk, food

1. Introduction

For decades, the livestock sector has focussed on improving the feed efficiency at animal level, i.e., the ratio of kg product output over kg resource input. This focus increased global production of animal-source food, but also the demand for land by the livestock sector. The current livestock sector uses about 70% of all agricultural land in the world (Steinfeld et al., 2006). The demand for especially animal-source food will further increase, and, therefore, intensify the claim on agricultural land, implying that land use efficiency (LUE) will become increasingly important for livestock production.

Life cycle assessment (LCA) is most commonly used to assess LUE of animal-source food products along the entire chain. A generally accepted indicator is land occupation. Current LCA results show that land occupation is in the same range for eggs (35-48 m^2/kg protein) and milk (33-59 m^2/kg protein) (De Vries and De Boer, 2010). They, however, also concluded that interpretation of current LCA results is hindered, because they do not include the environmental consequences of competition for land between humans and animals. Compared to diets of ruminants, for example, diets of laying hens, generally, contain more products, such as cereal grains, that humans could consume directly. Direct consumption of these cereal grains by humans is more efficient from a land use perspective than consumption of milk or eggs produced by animals fed with these cereal grains, because energy is lost during conversion from plant to animal product (Goodland, 1997).

One way to gain insight into the efficiency of land use by livestock, while correcting for this competition between humans and livestock, is to compute human-edible energy or protein conversion ratios (Wilkinson, 2011; Dijkstra et al., 2013). This protein conversion ratio, for example, represents the amount of protein in animal feed that is potentially edible for humans over the amount of protein in the animal product that is edible for humans. Wilkinson (2011) computed human-edible protein conversion ratios above 1, except for milk and suckler-beef production. Ratios above 1 are not sustainable because animals produce less human-edible protein than they consume (De Boer, 2012).

Another limitation of the land occupation indicator is that it does not account for the suitability of occupied land to cultivate food crops. Conversion ratios, as presented by Wilkinson (2011), do not include the fact that, for example, grass fed to dairy cows can be produced on land suitable for the cultivation of food crops. Some LUE indicators were proposed that correct occupied land for its production capacity, like the net primary productivity of potential biomass (NPP_0) (Ridoutt et al., 2012) and the ecological footprint indicator (Ewing et al., 2010; Borucke et al., 2013) or the quality of the soil (Milà i Canals et al. 2007b). All these methods, however, do not sufficiently account for the suitability of used land to produce food crops.

To efficiently feed an increasing and wealthier population, we need a novel method that overcomes above mentioned limitations in LUE assessment methods. The aim of this study, therefore, was to develop a novel method to assess efficiency of land use by livestock to produce human food. We illustrated our novel approach for the case of Dutch egg and milk production systems .

2. Methods

2.1. Life cycle assessment

We first used a regular LCA approach to assess land occupation for the production of eggs and milk. In this study, we used an attributional LCA to quantify the current status of LUE to produce egg and milk.

Land use assessment in an attributional LCA focusses on two main issues, i.e., the claim on land by production (e.g., the indicator land occupation) and the impact of used land on biodiversity and soil quality/life support function (Garrigues et al. 2012; Milà i Canals et al. 2007a). Indicators to quantify the impact on biodiversity and soil quality/life support function are still in development and it is not evident yet which indicators should be considered in LUE assessment of livestock systems (Helin et al. 2014; Koellner et al. 2013; Núñez et al. 2013), although some recommend to use soil organic matter (Milà i Canals et al. 2007b) as indicator for soil qualtiy (Food SCP RT 2013). Therefore, and because our focus was on the suitability of used land to produce food crops, we quantified only the claim on land by livestock production.

Land occupation in livestock production mainly results from cultivation of feed ingredients. Therefore, we took into account agricultural land use for feed production only.

2.2. Novel approach for land use efficiency assessment of livestock

Our approach for LUE assessment builds on current LCA calculations. A regular LCA sums occupied land for all feed ingredients per country of origin used for the production of, for example, a kg human-digestible (HD) egg or milk protein. Our approach sums the suitability of this occupied land per country of origin to directly produce HD protein from food crops. We chose HD protein production as parameter for LUE as livestock products contribute especially to the protein demand of humans (De Vries and De Boer 2010; FAO 2009). Protein digestibility was taken into account to correct for the quality difference of plant and animal protein. The concept behind our approach, therefore, is that to maximize LUE, HD protein production should be maximized per agricultural land unit. Our LUE ratio, therefore, was defined as:

$$\text{LUE ratio} = \frac{\text{HD prot livestock}}{\text{HD prot crop}} \qquad \text{Eq. 1}$$

where HD prot livestock is HD protein production from livestock (g HD protein/FU) and HD prot crop is HD protein production from most suitable food crop (g HD protein/FU).

For our case studies we chose kg HD egg or milk protein as FU. Current LCA studies, however, do not yet account for protein digestibility. Therefore, to compare our current land occupation results to values in literature, we expressed these results also in kg egg or milk protein. As the FU was expressed in kg HD livestock protein, the numerator of the LUE ratio was equal to 1000 g HD livestock protein.

The denominator of the LUE ratio was calculated according to Eq. 3.

$$HD\ prot\ crop = \sum_{i=1}^{n} \left(LO_{ij} \times \max_k \left(AY_{jk} \times Prot_k \times Dig_k \right) \right) \qquad \text{Eq. 2}$$

Where, LO_{ij} is current land occupation (m^2.yr/FU) for feed ingredient i and country j, AY_{jk} is average yield (g/m^2.yr) for country j and suitable food crop k, $Prot_k$ is protein content (%) for suitable food crop k, Dig_k is protein digestibility (%) for suitable food crop k and the max function is selecting food crop k with maximum HD protein yield for country j.

To calculate HD prot crop (Eq. 2), first, we identified the country of origin for all feed ingredients. Second, we assessed the suitability of these countries to grow food crops. Five major food crops were considered: wheat, rice (wetland and indica dryland), maize, potatoes (white and sweet) and soybeans. To identify country specific suitability for these food crops we used the GAEZ database on agro-ecological suitability and productivity (FAO and IIASA 2014). This database expresses country-specific suitability of land to grow food crops in a suitability index ranging from not suitable to very highly suitable. This suitability index is based on data of climate (e.g., wet day frequency, sunshine and temperature), crop requirements, prevailing soil conditions (e.g., pH, soil water holding capacity and total exchangeable nutrients), applied soil management, elevation and terrain slope, land cover and protected and administrative areas (FAO and IIASA 2014). If, according to the GAEZ legend, the suitability on country level was good, high or very high (i.e., suitability index >55), the land was considered suitable to produce the crop. Third, for every suitable food crop, the country-average yield in kg/ha was obtained from the FAOSTAT database (FAO 2014). Fourth, these yields were corrected for protein content and digestibility.

The LUE ratio corrects for both limitations in current LUE assessment methods. First, it corrects for differences in amount of HD plant products in livestock diets as these products, of course, are produced on land that has high alternative protein production from food crops, which makes the denominator larger. Second, we correct for suitability of used land to cultivate food crops. For example, when grass is grown on land that could have been used to grow food crops, the denominator becomes larger as well.

2.3. Case studies

We illustrated our approach for the cases of Dutch egg and milk production systems, because these systems yield similar LCA results for land occupation per kg of protein, whereas diets of dairy cows and laying hens differ in amount of HD plant products. As the diet of laying hens is relatively similar for different production systems, we took the most common egg production system in the Netherlands, i.e., a multi-tiered barn (Egg) (Dekker et al. 2011). Furthermore, we included two milk production systems in the Netherlands, i.e., milk production on peat soils (Milk Peat) and milk production on sandy soils (Milk Sand). We selected these milk production systems because of their differences in the suitability of land used for feed production to cultivate food crops.

Current land occupation for Egg was calculated based on data from Dekker *et al.* (2011), FeedPrint 2013.03 (Vellinga et al. 2013) and KWIN-V 2013-2014 (KWIN-V 2013). Data about the composition of the layer hen diet and the corresponding off-farm land use, feed intake and production values came from FeedPrint 2013.03. The egg price, slaughter price and round duration were obtained from KWIN-V 2013-2014.

When a hectare of land yields multiple products, land occupation needs to be allocated to these multiple products. Economic allocation was used to allocate land occupation between egg and meat production. Economic allocation means that occupied land is allocated proportionally to the economic value of the different products. Economic allocation was used for crops with an oil and a meal product also. Economic allocation values for these crops were based on FeedPrint 2013.03. According to FeedPrint 2013.03, residue co-products, like citrus- and beet pulp, maize glutenfeed and wheat middlings, had an economic allocation value of zero. The co-product straw was assumed to have an economic allocation value of zero.

Current land occupation for Milk Sand and Milk Peat was calculated based on data from the Dutch Farm Accountancy Data Network (FADN) database (FADN 2014) and FeedPrint 2013.03 (Vellinga et al. 2013). Data from the FADN database described average milk production, economic allocation percentages for milk as percentage of total revenues from milk and meat, on-farm land use with corresponding crop yields, total feed intake and the proportion of different feed ingredients in the milking cow diet over three years (2010-2012). To get a high contrast between production systems for dairy cows on peat and sandy soils, we selected FADN farms with >90% peat or sandy soils. Allocation rules were applied similarly to the laying hen case.

The HD protein production from food crops on land used for on-farm feed production was not solely based on the GAEZ and FAOSTAT databases. We assumed HD protein production from food crops on peat soils to be zero. Crop yields for wheat, i.e., 7,300 kg/ha.yr and white potatoes on sandy soils, i.e., 56,000 kg/ha.yr and wheat, i.e., 9,200 kg/ha.yr and white potatoes on clay soils, i.e., 50,000 kg/ha.yr were obtained from KWIN-AGV (KWIN-AGV 2012).

3. Results

Land occupation for egg production was about 26 m^2 per year per kg digestible protein (Table 1). On this land, which is spread all over the world, it is possible to produce about 2 kg of HD protein from food crops, resulting in a ratio of 0.5.

Table 1. Land occupation per kg human-digestible (HD) protein and LUE ratio for egg production (Egg) and milk production on sandy (Milk Sand) and peat soils (Milk Peat).

Product	**Land occupation (m^2.yr/kg HD protein)**	**LUE ratio**
Egg	25.8	0.5
Milk Sand	26.1	0.4
Milk Peat	31.6	1.7

Current land occupation was higher for Milk Peat than for Milk Sand, mainly due to higher grass and lower maize content in the diet on dairy farms on peat soils. On-farm maize yields were higher than on-farm grass yields, which led to lower land occupation for maize than for grass. The LUE ratio, however, strongly depended on the soil type. When a dairy farm was situated on peat soils, where hardly anything else than grass could be grown, the ratio was 1.7. This means that more HD protein from milk than from plants could be produced on the land where the feed ingredients are grown. When situated on sandy soils, however, the ratio was about 0.4, which means that production of milk on sandy soils is an inefficient way to produce HD protein.

4. Discussion

According to De Vries and De Boer (2010) the range in land occupation of egg production systems was 35-48 m^2.yr/kg protein. Our current land occupation value, recalculated to kg protein was below this range (25 m^2.yr/kg protein). This can be explained by the relative low feed conversion ratio (FCR) from FeedPrint. Besides, the range of De Vries and De Boer (2010) was based on two studies from 2006. Additional possible explanations, therefore, can be decreasing FCR and increasing crop yields over years. Furthermore, in the current study, only land occupation by feed production, i.e. agricultural land, was considered.

Current land occupation was 25 m^2.yr/kg protein for Milk Sand and 30 m^2.yr/kg protein for Milk Peat. According to De Vries and De Boer (2010) the range in land occupation of milk production systems was 33-59 m^2/kg protein. This range was based on studies from 2000-2009. Our relatively low current land occupation values, just as for the laying hen case, can possibly be explained by decreasing FCR and increasing crop yields over years. Two Dutch studies were used in the review of De Vries and De Boer (2010). These Dutch studies found a land occupation value for Dutch milk production of about 39 m^2/kg protein), which is on the lower side of the range as well (Thomassen et al. 2009; Thomassen et al. 2008).

Current land occupation was highest for Milk Peat, followed by Milk Sand and Egg, but still close to each other (25-30 m^2.yr/kg protein). This land occupation, however, was relatively similar for these livestock systems. This is in accordance with the findings of De Vries and De Boer (2010) that the amount of land required to produce 1 kg of protein is in the same range for eggs and milk. Opposite to our findings, Williams *et al.* (2006) found lower current land occupation values for milk production systems (33-35 m^2/kg protein) than for egg production systems (41-48 m^2/kg protein). This difference can possibly be explained by our relatively low FCR for egg production. Besides, compared to the ranges of De Vries and De Boer (2010) for land occupation of egg and milk production, Williams *et al.* (2006) found relatively high land occupation for egg production and relatively low land occupation for milk production.

The novel approach to assess LUE of livestock systems led to a change in conclusions of livestock systems on their LUE. Wilkinson (2011) found the same conclusions at animal level about the loss or gain of HD protein during the production of eggs and milk on peat soils. Our results for milk production on sandy soils, however, were different, because Wilkinson (2011) didn’t include the fact that, for example, grass fed to dairy cows can be produced on land suitable to grow food crops.

The land use inefficiency of dairy farms on sandy soils compared to dairy farms on peat soils could be explained mainly by the difference in suitability of both soils to grow food crops. Production capacity of land is not fully used for the case of milk production on sandy soils, as grass is grown on land that has high suitability to grow food crops. This shows that animals that convert non-HD protein, originating from land with low capacity to grow food crops, into HD protein efficiently, could contribute to increase global HD protein production. This finding is in agreement with the conclusion of Dijkstra *et al.* (2013) that we should fully utilize the ability of ruminants to convert non-HD products into HD products.

5. Conclusion

Our novel approach to assess efficiency of land use by livestock to produce human food led to a change in conclusions on the LUE of livestock systems. We found highest LUE for livestock production on land that has low suitability for HD protein production from food crops. These results show that animals that convert non-HD protein, originating from land with low capacity to grow food crops, into HD protein efficiently, could contribute to increase global HD protein production. This means that not only the actual type of land use (meadow, arable land, etc.) has to be taken into account in the assessment of efficiency of land use, but also its natural status (e.g. soil type), determining its suitability to produce food crops.

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Sustainability & Ethics – key issues for packaging trends

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ABSTRACT
Sustainability & Ethics is one of the five macrotrends identified by a group of experts at CETEA/ITAL - Packaging Technology Center of the Food Technology Institute that could drive the processes of innovation and development in the packaging area in the coming years in Brazil. LCA, in its simplest form, *Life Cycle Thinking,* is a powerful tool to be applied to continuous improvement of the existing processes as well as to lead to new products and process development. Many examples of the application of *Life Cycle Thinking* have been achieved with significant measurable reductions in energy, water, raw materials, emissions to water, and air and solid waste residues. The application of this tool for packaging and also for the food sector puts forward four guidelines to be pursued over the next decade: Optimization of the product/package system; Reuse & Recycling; Waste Management & Reverse Logistics and Credibility & Ethics.

Keywords: Sustainability, packaging, life cycle thinking, trends

1. Brazil Pack Trends 2020 report

In order to establish guidelines that could drive the processes of innovation and development in the packaging area in the coming years in Brazil , a group of experts at CETEA/ITAL - Packaging Technology Center of the Food Technology Institute - with over twenty years experience in research and consultancy, identified five macro trends that constitute challenges and opportunities that could encourage enterprises to think about the future possibilities for packaging: Convenience & Simplicity, Aesthetics & Identity, Quality & New Technologies, Sustainability & Ethics and Safety & Regulatory Issues (Sarantopoulos et al., 2012). The purpose of this article is to show the main topics concerning one of the macrotrends - Sustainability & Ethics, which was developed in Chapter 7 of the report Brazil Pack Trends 2020 (Mourad and Jaime, 2012). More than 70 documents including those on global trends were extensively studied which, using the authors´experience, gave rise to the main drivers of this chapter. So, the following sections of this article show the main points discussed in this chapter, but here applied to the food packaging sector.

2. Environment: a global issue

The first aspect addressed in this chapter is the global character that environmental issues have taken on. Globalization, which has intensified from the second half of the '80s, has transformed the environmental discussions from national to global. The social-related aspects refer to global society and not only the regionalized effects. Through globalization, humanity becomes aware of the risk of environmental degradation due to the potential destructive capacity of nuclear plants and contamination of air, water, soil and the food chain by chemical industries.

A new global conscience has gradually arisen as the transformations that the planet has been passing through, accentuated by climate changes that can be clearly seen in every continent, have been understood as consequences of the actions of mankind on nature (IPCC, 2007). The landslides that happened in 2011 in Teresópolis, in the State of Rio de Janeiro, due to a long rainy period, as well as the floods in Pakistan in 2010, the heat waves in France in 2003, the drought in Russia in 2010, the heat in the Alpine resorts in 2006 and the drought in USA in 2012, among other events, are reflexes of climate change. Since these events have been increasingly frequent, the need is urgent for all sectors of the economy, including the packaging sector, to contribute to the reduction in the emission of greenhouse gases and implement actions to reduce the anthropogenic impact on our system.

3. Life Cycle Thinking

As Life Cycle Assessment (LCA) studies measure various forms of environmental impact such as global warming, natural resource depletion, acidification, eutrophication and human toxicity, among others, it has been considered as one of the best instruments for the quantification of human action on the planet. The LCA tool itself is very complex and requires considerable time for completion. For this reason, partial LCA results for identification of opportunities for improvement, including all or almost all of the life cycle phases, have grown in recent years and are based on "life cycle thinking". These studies, although they cannot be classified as a "life cycle assessment", have been very useful for understanding the environmental interfaces of products and have lead to the development of products with lower environmental impacts.

Thus, with the greater maturity of environmental issues, this new more holistic view, "Life Cycle Thinking", has been used in the packaging context. This approach can be translated as "a continuous movement of rethinking packaging", considering the time when natural resources are extracted from nature for their production up to the final destinations of residues and emissions originated from these processes. Figure 1 illustrates how *Life Cycle Thinking* can be used for system optimization.

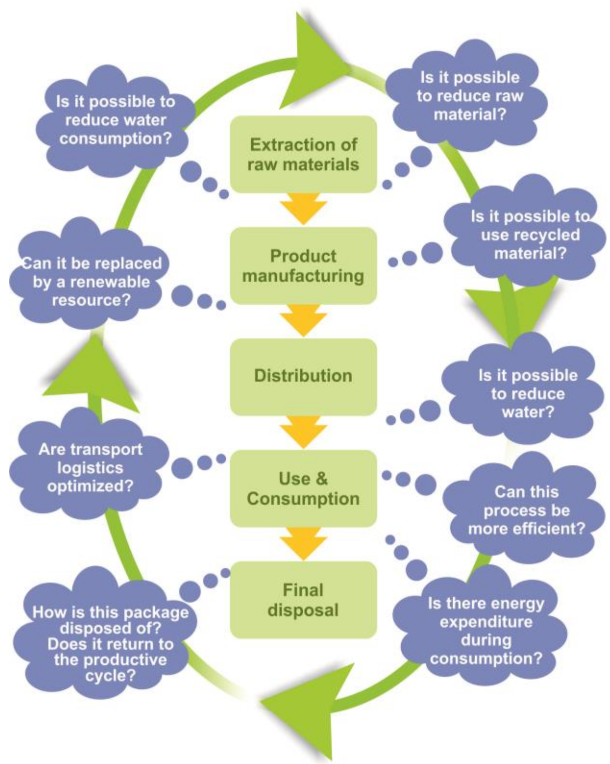

Figure 1. Schematic representation of Life Cycle Thinking.

Through these questions, asked at every phase of the product life cycle, *Life Cycle Thinking* is carried out. The registering, however, of the obtained reductions, must follow the LCA methodology strictly, so the calculated reductions are effectively proved on an internationally recognized scientific basis. It is also important to include analysis of possible trade-offs, ie, possible environmentally unfavorable points arising from the implementation of new processes, such as the increase of emissions to water when increasing the recycling rate or loss of packaged products when reducing the mass of some packages which become more fragile, and so on.

Packages have always been linked to environmental issues and their use is often questioned. It is important to remember that through the package it is possible to distribute food from its places of origin to multiple, often remote, points of consumption. Another aspect to remember is the importance of its correct specification. The use of inappropriate / undersized packages leads to food loss and consequently all natural resources extracted from the environment throughout their supply chain. Thus, the loss of food by failure or lack of packaging has negative consequences for the environment, many times greater than the impacts associated with the manufacture or disposal of packaging.

4. Guidelines for sustainable product/packaging design

4.1. Optimization of the product/package system

Life Cycle Thinking can currently be considered as one of the most important tools for the development of packaging and products that aim to become more sustainable. For packages, this concept means to "rethink the packaging associated with its lifecycle, challenging the limits of weight, shape, materials and accessories, without, however, compromising the integrity and product shelf life." When the wish to become less costly to the environment becomes a goal clearly defined, it reduces the weight of what is not essential; it goes to the limits of technical requirements, values the efficiency more than appearance and generates more responsible packaging options.

When the relation between product or service and the quantity of used package is optimized, the consumption of natural resources is indirectly reduced, such as oil, water, sand, coal and minerals, among others, and, consequently, the resulting emissions to the air, water and soil. This way, the optimization of materials should be one of the priorities in the search for systems with less environmental impact.

Many examples of the application of *Life Cycle Thinking* developed between 2009 and 2013 in a partnership consisting of a retailer (Walmart), manufacturers (Walmart suppliers) and a research institute (CETEA/ITAL) are described with significant measurable reductions in energy, water, raw materials, emissions to water, air and solid waste residues (Walmart Brasil, 2010 and 2011).

Coca-Cola Brazil invested in their organic product, Matte-Leão, which was produced in their new unit. Considered a "green factory", it has received the Leadership Energy and Environmental Design (LEED) certification, issued by the Green Building Council, which assures good planning with the construction and use of energy. Another important innovation was the printing of the life cycle of tea from mate herb on the package, playing an important role on educational and environmental consciousness. The package has been modernized by its clean design: 90% of the printing ink quantity has been reduced (Figure 2).

Figure 2. Organic Matte Leão, by Coca-Cola: 90% reduction of the printing ink. Source: Walmart, 2011.

The company Cargill has shown that it is possible to innovate on traditional products such as the Liza oils line (Figure 3), which, even maintaining the shape, thanks to a re-engineering work on the bottle and cap, a 10% reduction in the package weight was possible, which changed from 22 to 20 grams without significantly affecting its mechanical performance.

Figure 3. Example of optimization with packaging weight reduction. Source: Walmart, 2010.

Derived from technological investment, Danone has applied FOAM technology, which expands the plastic sheet used in the packaging, making it aerated, reducing its mass and, therefore, the consumption of natural resources. This change brought about a 9.4% reduction in weight of a Danoninho pot, without compromising its mechanical resistance (Figure 4).

Figure 4. Example of application of new technology for weight reduction. Source: Walmart, 2011.

4.2. Reuse & Recycling

On the scale of priorities for solid waste management, the minimization of waste generated by optimizing and reducing packaging systems should be encouraged first. Reuse is the second priority, followed by recycling, in third place.

Coca-Cola Brazil launched, in 2011, the 2.5 liter bottle containing 20% of post-consumer recycled PET resin food grade (PCR PET). There is, currently, a bottle-to-bottle recycling plant that was approved by the Brazilian Health Surveillance Agency (Anvisa). The recycled bottle (Figure 5) has the same characteristics as produced from virgin PET resin. In this process, the used bottles used are selected, crushed and washed by an efficient process of decontamination and recovery of the molecular weight of PET, removing contaminants at levels required by food contact packaging legislation. The clean material is then used in the manufacturing process of new bottles. The technology used for the purification of post-consumer PET was created by the company United Resource Recovery Corporation (URRC), from the United States. The use of post-consumer PET encourages the entire chain of collection of these packages, generating income for the street collectors.

Figure 5. Recycled PET bottle. Source: Walmart, 2011.

Example of New Can metallic packaging design, developed by Jiwoon Park and Kwenyoung Choi, has a format that allows for the reduction of volume by one-third of the original after consumption of the product by means of twisting and compression movements.

Figure 6. Example of a package that allows for volume reduction after product consumption. Source: YANKO DESIGN, 2009.

The Taiwanese company Panorama SOY Ink Co. Ltd. has developed inks based on recycled vegetable oils. The inks contain 45% of post-consumer oil, 21% of pigments and 34% of tar, do not contain organic volatile compounds, have good resistance to abrasion, as well as high gloss and good color stability. They dry faster, have better defined impression than traditional paints and can be applied on surfaces of paper or plastic.

4.3. Management of residues & reverse logistics

The establishment of the reverse logistics chain is not a simple task and involves many aspects. In fact, in cases where the use of packaging in a second production process already provides a financial return for the agents involved, as in the case of old corrugated paperboard boxes and aluminum cans, the return of post-consumer packaging already exists. However, in most cases, that chain needs to be created and established. It is very important, however, to realize that the creation of this chain does not occur spontaneously. If there does not exist a clear determination of the generating sector in returning these materials to the production cycle, or an appropriate destination, this chain cannot be established. In order to ensure that the reverse chain is real and can be maintained, it must be economically viable, which means that it must pay all the involved agents in an adequate way.

The Brazil Pack Trends report also presents a successful Tetra Pak case which established a reverse logistic chain for aseptic milk packaging in the country. This perception of the need to go beyond one´s own gates was perceived long ago by Tetra Pak, which is a great example of a company that was able to foster, encourage, and establish the reverse logistics chain of the aseptic packages after use. These containers of liquids are formed by the combination of three materials: cardboard, which gives rigidity and packing structure, alternating layers of polyethylene (PE), which protect the cardboard from external moisture and also constitute the primary contact material with the liquid beverage, and an aluminum foil (Al), which preserves the aroma and extends the shelf life of the product which reaches, in the case of milk for example, up to six months.

This multilayer material is currently separated from the common waste (the current recycling rate is 28%) and the cellulosic fiber content recovered in “hidrapulpers” present in paper recycling companies. The remaining residue consisting primarily of polyethylene and aluminum, is currently intended for the manufacture of PE/Al tiles and to the EET-Brasil Aluminum and Paraffin Ltd. company at Piracicaba. At the EET company, through a plasma process (~15,000ºC), high purity aluminum is obtained and the polyethylene is transformed into paraffin. LCA studies carried out by CETEA attest that recycling has environmental benefits, even considering the impacts of all these stages of the reverse chain.

The creation of this chain, exemplified in Figure 7, was strongly encouraged by Tetra Pak, which took nearly two decades to find the technology and partners that could make the reuse of aseptic packaging environmentally and economically viable.

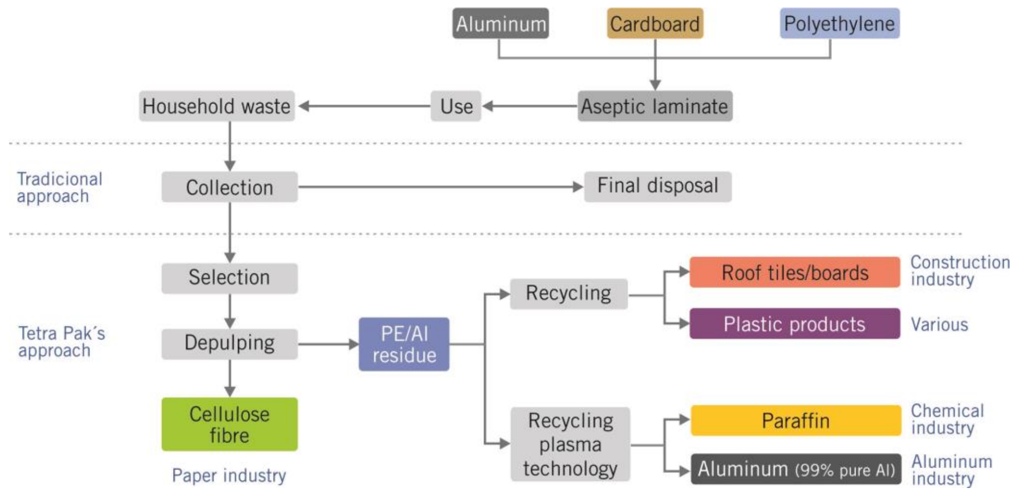

Figure 7. Productive and post-consumer aseptic package recycling flowchart organized by Tetra Pak. Source: Adapted from Osato et al, 2007).

The creation of these products increased the price paid for the collected post-consumer packaging by 79% between 2004 and 2007, which reached EUR 120 per ton and started to be segregated from the household waste.

5. Credibility & Ethics

With the globalization of environmental issues, the strengthening of social networks, the importance of the end consumer´s awareness and the involvement of the whole society in areas before treated only by experts, it is very important that environmental communication will be carried out according to internationally accepted metrics.

Many manufacturers have already noticed the need of having a better attitude to society, extending the responsibility concerning their products to beyond their gates. Such a movement has been called Extender Producer Responsibility (EPR). The publication of sustainability reports, according to the GRI Initiative, is a practice incorporated by several companies and the public sector that have a commitment to sustainable development and have already put into practice the EPR principles. The initiatives involve actions all along the production chain. GRI has been conceived so it is possible to communicate in a transparent way the entrepreneurial action of economic, environmental and social range, according to a method by which a company can be compared both internally and externally. GRI involves principles of balance, comparability, exactness, periodicity, reliability and clearness.

The credibility of the products has been attested using standardized environmental labels and self declarations; both volunteer and offer information about the environmental benefits of a service or product in general terms or one or more specific environmental aspects. The environmental performance certification of a product or service is a world-wide practice, with Germany being, in 1977, the first country to implement a National Program of Environmental Labeling for products, the Blue Angel. This kind of program has been used as a model for many other countries, becoming a strong worldwide trend, examples being: Canada (Environmental Choice), Japan (Eco Mark), United States (Green Seal), Nordic Countries – Denmark, Norway, Sweden, Finland and Iceland (The Swan), and Europe (Eco-Label), among others.

These programs belong to the Type I Environmental Labeling, established in the ISO 14024 and are also known as “green labels” or “ecolabels”. It is a voluntary methodology of certification and labeling for environmental performance of products and services, with a great importance for the implementation of environmental policies aimed at consumers, helping them to choose products less harmful to the environment. The “green label” is generally given by a national certification organ (third party) and is based on multiple criteria from studies of Life Cycle Assessment of a given sector, with focus on reducing the environmental impacts associated with the selected product category. The certification of cellulosic materials by internationally recognized agencies has become a more intense reality in Brazil and around the world, highlighting the label Forest Stewardchip Council (FSC), applied to the cellulosic packages sector. The Brazilian Program of Forest Certification (Cerflor), applicable nationwide, prescribed to the standards elaborated by ABNT and integrated into the Brazilian System of Conformity Assessment. Inmetro is another forest certification program used in Brazil, internationally recognized by the Program for the Endorsement of Forest Certification (PEFC), an international organization that unites systems of forest certifications all over the world. Both programs attest to the sustainable forest handling and the traceability of the custody chain, offering to the consumer the guarantee that the product follows criteria based on practices that promote biodiversity preservation, responsible use of forest resources and the maintenance of soil, air and water quality. They still analyze the practices of companies in the economic and social development of the region where they work. Another strong trend in the package sector is the adoption of the environmental self-declaration for companies, aiming at making public the environmental improvements obtained along a product or service life cycle. Although the self-declaration may offer flexibility and autonomy, as it does not demand the certification by a third party, the companies should make responsible declarations that can be verified and based on scientific rigor. This kind of declaration is called Type II Environmental Labeling and can be found standardized by the ISO 14021, which presents the policies for the use of texts, symbols, and graphs associated with the publicizing of product or service environmental improvements. Texts with vague or non-specific declarations, for example, “environmentally safe”, “environmentally friendly”, “Earth friendly”, “do not pollute” and “ozone layer friendly” should not be used. The consumer search for products with less environmental impact has helped the high investment by companies in selling with environmental appeal. This “green” market trend has stimulated companies to use the moment to associate their products with dubious and opportunist ecofriendly attributions, with no clear criteria that back them up in their environmental pretensions, or even to symbols and visual appeals that can induce the consumer to wrong conclusions about a product or service. Appeals that are presented as fake or induce the consumer to wrong conclusions about a product or service have been called *Greenwashing*. Aiming at describing, understanding and quantifying the growth of *Greenwashing* in the market, the Canadian environmental marketing consultancy TerraChoice has developed a research methodology by printing on the packages orientations about environmental self-declarations established by the ISO 14021 standard. In that report, such fake or dubious appeals were classified in seven categories, called The Seven Sins of *Greenwashing*. Recently, many manufacturers have launched products implying that they are obtained from natural sources. This does not certify that the product was manufactured using the lower environmental impact technologies available, often hiding trade-offs (sin 1) concerning processes with high loads of pollutants. It is possible to see many statements that attest carbon footprint reduction without, however, supporting information by a third party that can validate these statements is the committing of “no proof” (sin 2). To state that a product is compostable in places where an infrastructure for residue collection and composting plants do not exist, is to commit the “sin of vagueness” (sin 3). Statements such as "heavy metal free" or “CFC free” according to the TerraChoice description, can be classified “a sin of irrelevance” (sin 4), since the maximum limits of these components are determined by legislation.

Avoiding *Greenwashing* does not mean the expectation of a perfect product, but that the honesty, transparency and scientific base are founded on the environmental declaration. To combat *Greenwashing*, the Environmental Packaging International (EPI) in the United States released in October 2009 a database of sustainable packaging materials, in which the package suppliers submit data from which information about sustainability of their materials are reviewed and confirmed by a third party. Similarly, the ISO 14021, environmental self-declaration, does not accept “vague” and inaccurate texts either such as “eco-friendly”, “environmentally safe”, “eco responsible”, and so on.

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Benchmarking the environmental performance of specialized dairy production systems: selection of a set of indicators

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ABSTRACT

Benchmarking the environmental impacts of dairy production systems across the world can provide insights into their potential for improvement. However, collection of high-quality data for an environmental impact assessment can be difficult and time consuming. Based on a dataset of 55 dairy farms from different countries, this study aims to identify an effective set of indicators, that can explain most variation between farms and is relatively easy to quantify. Results show that global warming potential (GWP) per kg milk was highly correlated with nitrogen balance (NB) (r= 0.73, P<0.01) per kg milk, land use per kg milk (r= 0.58, p<0.01) and energy use per kg milk (r= 0.74, P<0.01). Correlations between phosphorus balance (PB) per kg of milk and other indicators, however, were low. Consequently, results of principal component analysis yielded two factors. The first factor consisted of GWP, NB, land use and energy use, all expressed per kg of milk, whereas the second factor contained PB per kg milk. We concluded, therefore, that GWP and PB can be used as proxies to benchmark specialized milk production systems across the world.

Keywords: life cycle assessment, nutrient balance, milk production, correlation analysis

1. Introduction

Milk is an important protein source in human diets. Around 57% of the protein content of an average European diet consists of livestock products, of which about one third is milk-derived (FAOSTAT 2013). The global demand for milk is expected to increase further due to population growth, rising incomes and on-going urbanization. Dairy production, however, has a major impact on the environment. The global dairy sector, producing both milk and meat, is responsible for about 30% of the anthropogenic greenhouse gas (GHG) emissions from livestock (Gerber et al. 2013). Moreover, the dairy sector contributes to resource scarcity (e.g. land and fossil energy) (De Vries and De Boer 2010).

Various environmental assessment methods have been adopted to evaluate the environmental performance of dairy production systems. Two methods are commonly used in the field of agriculture. The first method is based on a nutrient balance, and yields indicators such as nutrient use efficiency and the nutrient balance per unit product. The second method is based on life cycle assessment (LCA), and yields indicators such as global warming potential per unit product or land occupation per unit product (Halberg et al. 2005). Both methods, however, have their own drawbacks. On the one hand, a nutrient balance neglects certain environmental impact categories (e.g., land use, energy use); on the other hand, although LCA incorporates environmental impact categories which are overlooked by a nutrient balance, it is difficult and time consuming to collect all the data required to perform an LCA (Thomassen and De Boer, 2005). For cross-border benchmarking, the simplicity of the indicator sets can be very essential. The aim of this study, therefore, is to select an effective set of indicators from above mentioned methods, that can be used to benchmark the environmental performance of specialized dairy production systems.

Exploring correlations between various indicators can help stakeholders to identify an effective set of indicators to evaluate the environmental performance of dairy farms, i.e. a set of indicators which is relevant, needs a minimal amount of data and is understandable (Lebacq et al. 2013). Previous studies have focused mainly on correlations between indicators within LCA. Their findings suggested that the number of indicators may be reduced because of strong correlation between some indicators (Berger and Finkbeiner 2011; Huijbregts et al. 2010; Laurent et al. 2012; Röös et al. 2013). So far, however, correlations between various indicators derived from a nutrient balance approach and LCA have not been explored within dairy production systems.

The objective of this study is to identify an effective set of indicators to benchmark the environmental performance of dairy production systems. We, therefore, explored correlations between indicators derived from a nutrient balance and from an LCA and see whether we can have a set of simplified indicators that can be used as proxies to benchmark environmental performances of dairy farms.

2. Material and methods

2.1. Data

Data was collected from 55 specialized dairy farms from Dairyman which is a project in the INTERREG IVB program co-funded by the European Regional Development Fund (Dairyman 2010). Environmental performance of these farms has been determined based on different indicators using data of the year 2010. We defined specialized farms as farms that have less than 5% non-dairy purpose animals, and less than 10% of their agricultural area in use for non-dairy purpose activities. The amount of energy, land and fertilizers used for non-dairy purposes was based on farmers' estimates and excluded from the data set. These 55 dairy farms are from different countries and regions (i.e. Netherlands, Ireland, Belgium (Flanders, Wallonia), France (Brittany), Germany (Baden, Württemberg) and Luxembourg).

2.2. System boundaries

As illustrated in Figure 1, we defined different system boundaries for different methods (i.e. nutrient balance and LCA). For the nutrient balance, we examined the farms' performances both at farm and partial chain level. At the farm level, we considered inputs (e.g. concentrates, roughage, animal) and outputs (e.g. milk, meat, crops) per farm, while the farm itself was considered as a black box. At the partial chain level, the system boundary was from cradle-to-farm gate. In addition to nutrient losses at the farm, nutrient losses during the production of purchased feed products were considered. Production of other farm inputs were excluded because their contribution to nutrient losses was assumed to be negligible. For both farm and partial chain levels, manure outputs of the farm were deducted from the input of the system. Stock changes (defined as final stock – initial stock) of the concentrates, roughages and fertilizers were taken into consideration during the computation processes.

For LCA, the system boundary was from cradle-to-farm gate. System boundaries, therefore, included the primary production of farm inputs, and all processes on the farm, such as manure management, milk and feed production. Pesticides and water usage were not considered due to lack of data. Capital goods (buildings and machinery) were not considered because their contributions to the environmental impact of dairy farming were assumed to be negligible (Cederberg 1998).

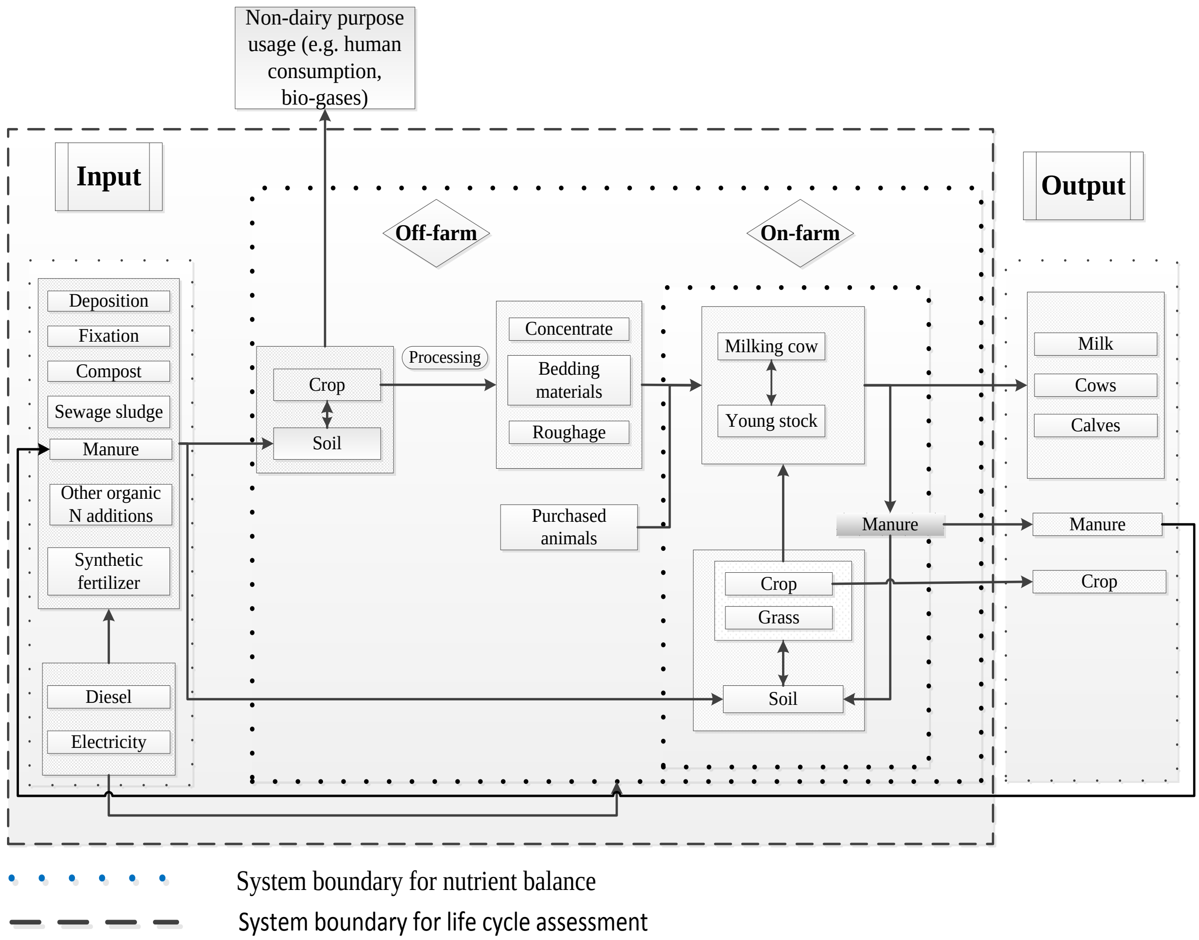

Figure 1. System boundaries

2.3 Nutrient balance

Nitrogen (N) and phosphorous (P) are essential nutrients for plant and animal production. The increased use of N and P for food production, however, also increased emissions of N and P compounds to the environment (Amon et al. 2011). In this study, based on N and P content in all inputs, outputs and stock changes of the systems, we computed N balance (NB) and P balance (PB) for each of the 55 dairy farms both at the farm level and partial chain level. The partial chain level nutrient balance was calculated by summing up the nutrient balance at farm level and the nutrient balance related to production of purchased feed products. Nutrient balance was defined as input-output and was expressed as nutrient balance / kg FPCM (fat-and-protein-corrected milk, i.e. milk is corrected to a fat percentage of 4.0% and a protein content of 3.3% using the formula: FPCM (kg) = Milk (kg) x [0.337 + 0.116 x Fat (%) + 0.06 x Protein (%)] (Product Board Animal Feed 2008)). Table 1 shows relevant inputs, outputs and calculation processes.

Table 1. Nutrient use efficiency for nitrogen and phosphorus

Elements	Calculation[4]	References
Input:		
Purchased animals	Q x nutrient content of animals	Raison et al. 2006
Purchased roughage[1]	Q x nutrient content of roughage	Dairyman 2010
Purchased concentrate	Q x nutrient content of concentrates[5]	Dairyman 2010
Purchased mineral fertilizer	Q x nutrient content of mineral fertilizer	Dairyman 2010
Purchased organic fertilizer	Q x nutrient content of organic fertilizer	Dairyman 2010
Atmosphere deposition	Average value of relevant region	EMEP 2007
Plant fixation[2]	Average value of relevant region	Raison et al. 2006
N&P input for producing feed[3]	N&P input for producing feed	Vellinga et al. 2013
Output:		
Milk output	Q x nutrient content of milk	Dairyman 2010
Animal output	Q x nutrient content of animals	Raison et al. 2006
Organic fertilizer output	Q x nutrient content of organic fertilizer	Dairyman 2010
Plant outputs	Q x nutrient content of plant	Raison et al. 2006

[1] We selected five most commonly used roughages in the dataset of Dairyman, i.e. grass silage, maize silage, hay, wheat straw and alfalfa.
[2] Plant fixation is only applicable for nitrogen
[3] Applicable for nutrient balance calculation at the partial chain level
[4] Q= actual quantity of product purchased or sold, obtained from Dairyman (in kg or numbers)
[5] Composition of the concentrate is based on the information provided by Dairyman

2.4. Life cycle assessment

The impact categories we examined are climate change, land use, and fossil energy use. The functional unit is FPCM. The computation for climate change is described in details in the following subsection.

Climate change. We used global warming potential (GWP) per kg FPCM as an indicator to assess the impact of milk production on climate change. GWP included emissions of the greenhouse gases (GHGs): CO_2, CH_4 and N_2O and was expressed in kg CO_2 eq per kg FPCM. Emissions of GHGs were summed according to the following equivalence factors (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al. 2007).

We computed on-farm emissions of GHGs according to IPCC guidelines (IPCC 2006), generally at Tier-2 level. On-farm emissions included CO_2 emissions from lime and urea application, and combustion of energy resources, such as diesel; CH_4 emissions from enteric fermentation and manure management, and N_2O emissions from manure management and managed soils.

To compute CO_2 emissions from lime and urea application, we used the default emission factor of 0.12 for limestone, 0.13 for dolomite and 0.2 for urea as provided by IPCC (2006) Tier 1. To compute CO_2 emissions from on-farm energy use (i.e. fuel), we used emission factors from Eco-invent (2010).

Enteric CH_4 emission was determined as a function of the average gross energy (GE) intake of the dairy herd and a methane conversion factor of 6.5% (IPCC, 2006). Average GE intake of the dairy herd was calculated based on the total net energy requirement of the herd (i.e. energy required for growth, maintenance, activity, and milk production) and energy availability characteristics of feeds. CH_4 emissions from manure management were calculated based on the amount of volatile solids in manure and manure management system (IPCC, 2006).

During storage, treatment and application of manure, N_2O is emitted via direct and indirect pathways. Direct N_2O emissions from manure storage and treatment were calculated by multiplying the total amount of N excretion within a certain manure management system with an emission factor for that type of system (IPCC, 2006). Direct N_2O emissions from manure application were calculated by multiplying the amount of N in manure applied to the field with the default emission factor of 0.01 (IPCC, 2006). Indirect N_2O emissions were based on the amount of N volatilized in the form of NH_3 and NO_x, and the amount of N leached in the form of NO_3 (IPCC, 2006). In addition, direct and indirect N_2O emissions from application of synthetic fertilizer, crop residues, and from urine and dung deposited on pasture during grazing were included and based on IPCC (2006) calculation rules.

Off-farm GWP was calculated based on GHG emissions from production and transportation of purchased feed (e.g. concentrates, roughages, by-products and from production of synthetic fertilizer. Concentrates were categorized into four categories: cereals (i.e. wheat, barley, triticale, oat, and corn), rapeseed meal, soya meal, and other. GHG emissions related to production of purchased feed products were based on Vellinga et al. (2013). GHG emissions related to production of fertilizers were based on Ecoinvent (2010).

Land use. This impact category examines the area of land (m^2) used to produce one kg FPCM. For on-farm land use, we considered agricultural area used for dairy purpose. For off-farm land use, we considered land used for cultivation of concentrate ingredients, roughage and by-products (Vellinga et al. 2013), and for the production of fertilizers (Ecoinvent, 2010).

Energy use. On-farm energy use relates to milking of the dairy cows and on-farm feed production, and is based on Dairyman (2010). Off-farm energy use related to the production of purchased feed products was based on Vellinga et al. (2013); energy use related to the production of fertilizers was based on Ecoinvent (2010). We assumed that energy used in this study are electricity and diesel.

2.5 Statistical analysis

Principle component analysis (PCA) was applied to characterize the indicators to get an efficient set of indicators that can explain most of the variation. We applied PCA on seven indicators at the chain level i.e. NB, PB, GWP, land use and energy use. Spearman Rho's correlation analysis was conducted due to the fact that most variables are not normally distributed. After that, we did Kaiser-Meyer-Olkin test (KMO>0.5) and Bartlett's Test of Sphericity ($p<0.05$) in order to assure the adequacy of data sampling and suitability to conduct the PCA. We selected factors based on the rule of eigenvalue > 1. All statistical analyses in this study were performed using the software SPSS (SPSS 2011).

In order to interpret correlations identified from correlation analysis, we used the criterion in Cohen (1998, as cited in Röös et al. 2013), where defines r=0.1-0.29 (weak correlation), r=0.30-0.49 (medium correlation) and r=0.5-1 (strong correlation).

3. Results

3.1. Nutrient balance

As shown in Table 2, for nitrogen, the average farm balance/kg FPCM is 16g N, whereas the average chain balance/kg FPCM is 19g N; for phosphorous, the average farm balance/kg FPCM is 1g P, whereasthe average chain balance/kg FPCM is 2g P.

Table 2. Results of nutrient balance

Indicator	Unit	Mean	SD	Maximum	Minimum
Nitrogen					
Farm balance	g N / kg FPCM	16	8	34	4
Chain balance	g N / kg FPCM	19	7	37	6
Phosphorous					
Farm balance	g P / kg FPCM	1	1	3	0
Chain balance	g P / kg FPCM	2	1	8	1

3.2. Life cycle assessment

According to Table 3, producing one kg FPCM contributes to climate change (1.31 kg CO_2 eq), meanwhile, it uses 1.29 m^2 land and 2.92 MJ energy. On-farm processes have a larger impact on climate change and land use than off-farm processes. However, off-farm processes use more energy than the on-farm processes. Finally, variation between farms is greater for on-farm processes than for off-farm processes.

Table 3. Results of LCA

Impact category	Unit	Mean	SD	Maximum	Minimum
Climate change	kg CO_2 eq / kg FPCM				
On-farm		1.08	0.25	1.76	0.75
Off-farm		0.23	0.81	0.54	0.04
Total		1.31	0.29	2.08	0.94
Land use	m^2 / kg FPCM				
On-farm		0.82	0.38	2.23	0.32
Off-farm		0.47	0.21	1.48	0.11
Total		1.29	0.45	2.80	0.76
Energy use	MJ / kg FPCM				
On-farm		0.53	0.15	1.11	0.21
Off-farm		2.39	0.69	4.74	1.28
Total		2.92	0.74	5.26	1.49

3.3. Statistical analysis results

Results of correlation analysis are reported in Table 4. NB shows low correlation with PB (r= 0.04). GWP shows strong correlations with NB (r= 0.73, P<0.01), land use (r= 0.58, p<0.01), and energy use (r= 0.74, P<0.01). Energy use shows strong correlation with NB (r= 0.57).

A positive nutrient balance can cause emissions like N_2O which is a major contribution to the GWP, therefore GWP shows a strong positive correlation with NB. Land use and GWP are both expressed per kg FPCM. A possible explanation for the correlation between land use and GWP is that farms that are less efficient will have a lower milk production per ha (i.e. a higher land use), but also higher losses and, therefore, higher GHG emissions per kg FPCM. In addition, during production and use of energy, there are emissions like CO_2 and N_2O. All these emissions can increase GWP, therefore a high positive correlation is found between GWP and energy use.

The results of Kaiser-Meyer-Olkin (KMO) test and Bartlett's Test confirmed the suitability of conducting a PCA (principal component analysis). Sampling size is adequate (KMO= 0.655) and Bartlett's test of sphericity is significant (p= 0.000). Two factors were extracted from PCA, where NB, GWP, land use and energy use are grouped into one factor and PB is grouped into the second factor. The first factor can explain 58% variation existing in the data, and by also considering the second factor, total 79% variation existing in the data can be explain.

Table 4. Results of correlation analysis

	N balance	P balance	GWP	Land use	Energy use
N balance[1]	1	0.037	0.728^{**}	0.359^{**}	0.567^{**}
P balance[1]	0.037	1	-0.220	0.082	-0.211
GWP	0.728^{**}	-0.220	1	0.584^{**}	0.739^{**}
Land use	0.359^{**}	0.082	0.584^{**}	1	0.498^{**}
Energy use	0.567^{**}	-0.211	0.739^{**}	0.498^{**}	1

** Correlation is significant at the 0.01 level (2-tailed)
* Correlation is significant at the 0.05 level (2-tailed)
[1] N balance and P balance here are both at the chain level.

4. Conclusion

Results of the correlation analysis show that global warming potential (GWP) per kg milk was highly correlated with nitrogen balance (NB), land use, and energy use. Correlations between phosphorous balance (PB) and other indicators, however, appeared to be low. Consequently, results of PCA yielded two factors. The first factor consisted of the indicators GWP, NB, land use and energy use, whereas the second factor contained PB. Therefore, we concluded that GWP and PB can be used as proxies to benchmark specialized milk production systems across the world. Future studies might include other environmental indicators, such as eutrophication potential and acidification potential, to further explore possibilities for identifying an effective set of indicators to benchmark environmental performance of dairy production systems.

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Lessons Learned teaching a Massive Open Online Course on the Sustainability of Global Food Systems

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ABSTRACT
In the summer of 2013, we delivered a Massive Open Online Course (MOOC) exploring the *Sustainability of Food Systems: A Global Life Cycle Perspective*, which saw thousands of people from 138 countries enrolled in the course. We focused on increasing awareness of the complexity and global reach of modern agriculture and food production systems while enabling students (i.e., consumers) to understand the implications and impacts of their personal/household dietary decisions. The course included full cradle-to-grave consideration of food production; from the seeds that may have been genetically modified, through the choice of organic or synthetic fertilizer application, to the nutrient and calorie waste from poor handling or food spoilage. Through the interactive, discussion-based nature of the course, the students learned at least as much from stories and experiences shared by each other as from the video lectures and readings. As we hoped, many students left the course with an understanding of the substantial environmental impacts from activities throughout the life cycle of food production, and the importance of informed decision-making at the consumer level.

Keywords: teaching, Massive Open Online Course (MOOC), global food systems, sustainable food systems.

1. Introduction

Food systems represent a strong opportunity for those interested in life-cycle assessment (LCA) to affect positive change in reducing the environmental burden of human activities. Agricultural activities cover 38% of global land area (The World Bank), and are responsible for 10 to 12% of global anthropogenic greenhouse gas emissions (Smith et al. 2007) (2011 and 2005 figures, respectively). The supply chains that depend on agriculture are complex, but this complexity means there are many possible intervention points. In the spirit of the adage “an ounce of prevention is worth a pound of cure”, changing the behavior of the final consumer of food and agricultural products has a magnified impact on the agricultural practices at the start of the supply chain. The most effective way to achieve this change may be through education.

From June through August 2013, we helped thousands of students become better educated about global food systems through a Massive Open Online Course (MOOC) on the Coursera platform entitled *Sustainability of Food Systems: A Global Life Cycle Perspective*. A MOOC is free for everyone, anywhere so long as they have an Internet connection; there are no prerequisite courses or degrees, and the students are essentially free to interact with the course materials and each other as much or as little as they wish. Our course saw enrollment from 138 countries, and participation by more than 15,000 students.

Like many extension and outreach-focused efforts from the life-cycle assessment community, this course aimed to increase awareness of the complexity and global reach of modern agriculture and food production systems while enabling students (i.e., consumers) to understand the implications and impacts of their personal/household dietary decisions. Course content was organized around exploring the answers to 15 overarching questions. These included: How and where do we grow our food?; How do we choose what we eat?; and, How does what we eat affect our environment? As illustrated in Figure 1, we began at a personal level in discussing the nature of food and our individual relationships with it, then expanded our scope of investigation in order to illustrate the global nature of modern agricultural supply chains, then looked at the health and environmental impacts associated with this global supply chain, and finally returned focus to the individual in exploring how our personal or household choices could contribute to a more sustainable food system.

In this paper, we will discuss course organization, highlight some of the course topics most relevant to the food LCA community, and reflect on the key outcomes from this experience.

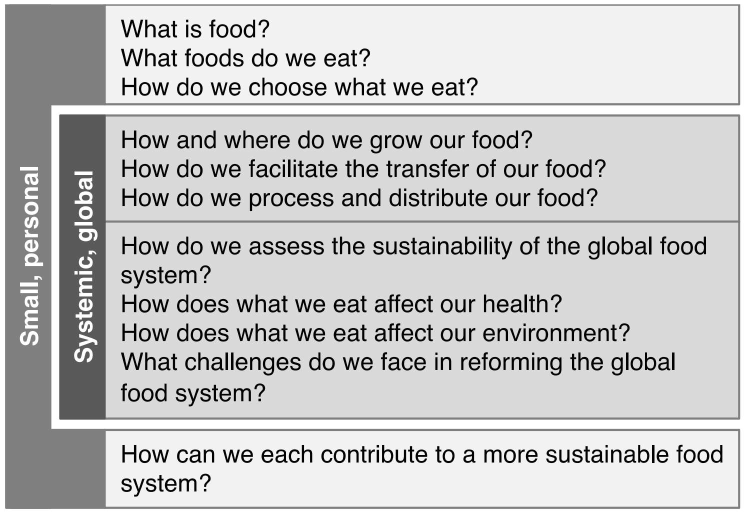

Figure 1. The complex global nature of agricultural supply chains and environmental impacts was the focus of our course, but we bookended the content with smaller scale, personal considerations so that students realized they could still affect change despite being a single actor in the system.

2. Materials and Methods

Course content was delivered in two ways: through readings, and through recorded conversations between Prof. Jason Hill and topic experts from the University of Minnesota. To keep the course ‘open’, readings were required to be accessible to anyone, anywhere, with zero cost. This ruled out using certain published journal articles, or requiring that students purchase any textbooks. It did not, however, rule out small extracts from books or articles as per fair use rules under copyright law. A complete list of readings is included in Table 1.

Table 1. Reading materials used in *Sustainability of Food Systems: A Global Life Cycle Perspective*.

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One of the key challenges of teaching in a MOOC environment is how to assess students' comprehension of the materials and concepts presented. Due to the sheer number of participants, it is not possible for student responses to an essay prompt to be assessed by a professor or teaching assistants. Multiple choice or filling the blank type questions are implementable, but only go so far in assessing student understanding of content. This latter assessment is problematic in a course such as this that requires students to incorporate new knowledge into what they already know about food systems. Our approach, then, was to get the students to engage with each other and discuss topics through the discussion forum on the course website (see Figure 2 for a screenshot). Through these interactions, the students learned at least as much from shared stories and experiences as they did from the video lectures and readings. As instructors, we also learned from these exchanges; stories from students in developing countries about ingenious local production practices, or from students directly involved in food aid assessment add color and context to more academic publications on which we base our understanding of the topics we teach.

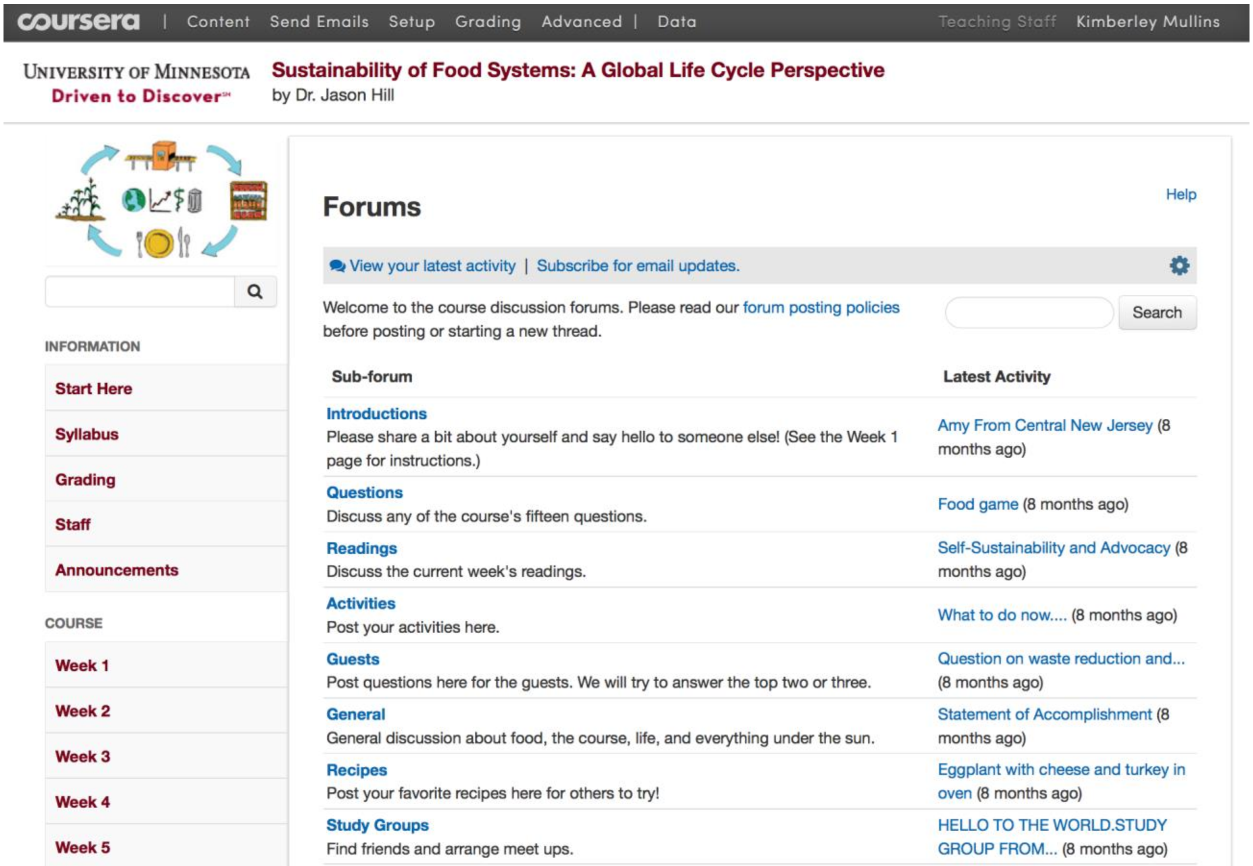

Figure 2. A screenshot of the Forums section of the course on the Coursera platform. Sub-forum headings directed the students towards the appropriate places to post questions, respond to assignment prompts, etc.

To introduce students to the beginning of the food supply chain, we discussed where our food originated historically. This issue is important in considering the sustainability of the global food system, as the plants and animals we consume now depend on the domestication decisions that were made many years ago. One assigned reading highlighted the fact that the domestication process is, in fact, ongoing in Africa (Pye-Smith 2009). We asked the students to pick a recent meal they had eaten, and determine to which regions in the world the basic ingredients of their meal were native. For example, if one had eaten spaghetti and meat sauce (wheat, cow, tomato and basil), the list would include the Eastern Mediterranean, the Andes, and India. The course materials and the exercise encouraged the students to reflect on how the availability of food has changed over time, and to consider the idea of “eating locally”, popular among those wishing to reduce the environmental impacts of their diets, in a new context.

The question of where our food originated was followed by a discussion of how our food is grown today. Prof. Paul Porter of the Department of Agronomy and Plant Genetics at the University of Minnesota discussed the myriad of agricultural practices currently undertaken around the world, highlighting that many practices in the world do not resemble the large-scale agriculture practiced in the midwestern United States. Of course, different practices characterized by labor and chemical input are associated with different yields. We had the students explore which countries produced which crops, and with what yields, using an online database from the UN’s Food and Agricultural Organization called FAOSTAT. This database is free and easy to query, important characteristics for a diverse student population.

A key step in the food supply chain is the conversion of these raw, basic ingredients into what is termed ‘processed food’. This can include anything from pasta to a ready-made frozen meal. This step has become an increasingly important one in developed countries, though the term ‘processed food’ is often used with negative connotations. Students read an article discussing the social and dietary impacts of major multinational food companies moving into Brazil (Monteiro and Cannon 2012). The students then heard Prof. Gary Reineccius of the Department of Food Science and Nutrition at the University of Minnesota describe food scientists' innovative work in food processing to improve the sustainability of the global food system. For example, food scientists can make whole grains more palatable to the general public through the calculated use of polyphenols, a natural plant extract that stops a chemical reaction during baking that produces an undesirable bitter flavor. The unsophisticated response is simply to add sugar to mask the bitter taste. The result of this food science development is that more whole grain low-calorie foods are consumed.

Environmental impacts lie at the heart of assessing the sustainability of the global food system. Prof. Stephen Polasky of the Departments of Applied Economics and Ecology, Evolution, and Behavior presented the concept of ecosystem services to the students. We felt it important to acknowledge that sustainability depends on more than reducing greenhouse gas emissions, but rather the natural systems upon which our food supply depend also influence biodiversity, water quality, air quality, and nutrient flows. A key point made by Prof. Polasky is that the food services we want from our natural systems, namely the profitable yield of food, can be at odds with the services we agree are beneficial but are not easily bought or sold, such as water remediation.

Some students were aware of these types of impacts or benefits because of the various food-labeling standards in place around the world. A discussion of supply chain impacts from Prof. Tim Smith, Director of the Northstar Initiative for Sustainable Enterprise, and a reading on "greenwashing" (Dahl 2010) forced the students to more carefully consider the claims made by labeling organizations and their eco-labels. His central point was that there is an important tradeoff between how rigorous and scientific the labeling requirements are and how many companies will actually engage in the process to receive the label. To reinforce this discussion, we asked the students to investigate two products that bore some claim to sustainability, with the expectation that students would find the requirements to earn the label, or the research and testing done to support an in-house claim of sustainability, vary widely among companies and labeling organizations. A handful of students were able to deliver what amounted to impressive investigative reporting to the rest of the class.

We wanted to make sure the students left the course feeling empowered to affect positive change and hopeful that change is possible, which can seem challenging in the face of a complex global system full of problems. To do so, we discussed broadly how we can provide food more sustainably, and how the individual consumer can contribute to a more sustainable global food system. The students read and heard the strategy of Prof. Jon Foley, Director of the Institute on the Environment, to feed a growing population more sustainably by increasing yields in low-yielding areas of the globe, increasing the use of modest technology options to improve the ratio of calories output to units of water or fertilizer input to an agricultural system, to reduce global meat consumption, and the reduce food lost to waste (Foley 2011).

3. Learning Outcomes

At the end of the course, we asked the students to reflect on what they had learned, and how they could apply their new knowledge in their own lives to improve the sustainability of the global food system. Some key themes emerged in reviewing student responses to the prompt. One of the most common responses given by students was their plan to reduce meat consumption, or at least reduce beef consumption, in light of the negative impacts of a global food system that relies heavily on the production and processing of animal protein.

Many students also left the course feeling better able to find further information on the topic we studied, or new topics we were not able to cover given the introductory nature of our curriculum. This result came from the introduction of tools, such as the FAOSTAT database, resources, of reputable organizations or scholars in the field of food systems and life-cycle assessment, and, simply, from a newfound vocabulary the students could leverage to find reliable information online more quickly.

From a pedagogical standpoint, we learned that emphasizing student interaction in the forums had many advantages, but it was not without drawbacks. Like tends to attract like, and in many cases this lead to a small community within the class that could share resources on specific interests such as vermiculture, managing dietary restrictions related to illness, or the challenges of food aid policy. Unfortunately, on some occasions these communities simply propagated misinformation. Perhaps the strongest such case was a completely erroneous discussion that arose about how using microwaves to heat food causes cancer because radiation affects DNA. We interjected with a post carefully crafted to explain the important differences between ionizing and non-ionizing radiation, and how microwaves employ only the latter, but it went completely unheeded by the discussion participants. Unfortunately, some deeply held convictions and prejudices were not shaken, and the nature of MOOCs made this task all the more difficult.

4. Conclusions

Agriculture, a global system expanding by necessity to feed an increasing population, is associated with many substantial environmental and health issues and changes in human behaviour are needed. This MOOC presented us with an opportunity to increase the awareness of the individual decision maker with regards to the complex and global nature of the modern agricultural system, and the challenges present, along with the resulting opportunities to improve sustainability. Based on participation and the ensuing robust discussions, we believe our efforts were successful.

In addition to being more aware, many students left the course believing that their individual efforts could have an impact on upstream activities in the supply chain. Some declared that they would change their behaviors; others argued that they could not, citing lifestyle pressures and other challenges. Some gained a confirmation of their current choices. The fact that such attitudes were expressed indicated successful engagement.

It was beyond the scope of this (or any) MOOC to assess the true impact on behavior; that is, will students actually behave differently? That said, we feel confident that our efforts have captured their attention and given them cause to consider the course of action they might take.

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The World Bank Agricultural & Rural Development Data.

Comparative life cycle assessment of five different vegetable oils

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ABSTRACT

The purpose of this work was to evaluate from cradle to gate the environmental performance of five vegetable oils: palm oil, soybean oil, rapeseed oil, sunflower oil and peanut oil, using a consequential approach and including indirect land use change. The impact assessment focused on greenhouse-gas (GHG) emissions, land use and water consumption. For GHG emissions, rapeseed oil and sunflower oil were the best performing ones, followed by soybean oil and palm oil and with peanut oil as the oil with the highest impact. Focusing on water consumption, sunflower oil was the oil with the smallest impact, followed by rapeseed oil, palm oil and soybean oil, and with peanut oil as the oil with the largest contribution. Regarding land use, palm oil and soybean oil were the oils associated with the smallest contribution, followed by rapeseed oil, and with sunflower oil and peanut oil as the oils with the largest net occupation of land.

Keywords: Palm oil, Soybean oil, Rapeseed oil, Sunflower oil, Peanut oil.

1. Introduction

The aim of this study was to evaluate the environmental impacts of a number of the major vegetable oils: palm oil, soybean oil, rapeseed oil, sunflower oil and peanut oil. Different vegetable oils systems are associated with different quantities of co-products, mainly oil meals, which are used as animal feed. When studying market responses related to changes in demand for the different oils and when substituting different oils, it is a challenge to address the interactions among oils and with the feed markets. The studied product systems were identified using a systems perspective approach where likely market responses and substitution effects are considered.

Previous research on comparative life cycle information on vegetable oils is relatively limited. Examples are Arvidssona et al. (2013) and Schmidt (2010). A larger number of studies exist within the field of biodiesel (e.g. Menichetti et al. 2009), which are most often limited to focus only on GHG emissions compared to mineral diesel (Mentena et al. 2013).

2. Methods

2.1. Goal, scope and functional units

The purpose of the study was to obtain environmental information on different major vegetable oils, to provide decision support for situations where different vegetable oils can be used, i.e. where the oils are substitutable. A functional unit of one ton refined (neutralized, bleached and deodorized; NBD) vegetable oil at refinery gate was used.

2.2. Consequential life cycle inventory modelling

The consequential modeling principles are comprehensively described in Ekvall and Weidema (2004) and Weidema et al. (2009). This approach was consistently applied throughout the study. The attributional approach would fail to comply with the purpose of the study, which focuses on predicting the environmental consequences of choosing different oils.

Production of refined vegetable oils is characterized by being associated with several by-products where the major ones are the oil meals from the oil mills and free fatty acids (FFA), which are a by-product from the refining process. Both the oil meals and FFA are used as animal feed. Schmidt at Weidema (2008) identified two major segments of the global generic animal feed market; namely feed protein and feed energy. Therefore a change in supply of oil meals and FFA will substitute these two products in proportion with their protein and energy content. Since the attributional approach only estimates/approximates the effects from downstream processing of by-products and product substitutions by applying an allocation factor on the upstream effects, this approach does not reflect actual cause-effect mechanisms in the market. The technique for performing the substitution calculations for vegetable oil systems is demonstrated in several studies, e.g. Dalgaard et al (2008) and it is also implemented in ecoinvent v3 (ecoinvent Centre 2013).

2.3. System boundaries

The inventories were established to represent 2011. The production functions for oil mill and refinery operations were regarded as being relatively constant over time, so data for 2005-2010 were used to represent 2011.

The study followed the same cut-off criteria as the ecoinvent v2.2 database (ecoinvent 2010). This implied that inputs of services (such as cleaning, accounting, lawyers, marketing, business travelling), research and developing (laboratories, equipment, offices etc.), and overhead (overhead energy, office equipment etc.) were not included. Further, the use of pesticides was not included in the study. Indirect land use changes (iLUC) were included (see section 2.3).

The oils were inventoried from cradle to gate (at refinery). Each of the five oil product systems included three product stages: 1) oil crop cultivation, 2) oil mill, and 3) refinery. Generally, the oil mills supply crude oil and oil meal (feed energy and feed protein) and the refineries supply refined oil and free fatty acids (feed energy). However, the palm oil system includes an additional step, since the kernels from the fresh fruit bunches are sent to another oil extraction step. As an example, Figure 1 shows the process diagram and mass balance for the palm oil and rapeseed oil systems. When solving the inventory problems related to multiple product output systems using substitution, it was necessary to identify which one of the co-products was the determining one. The latter is defined as the one for which a change in demand leads to a change in supply. In short, for all oils, except the soybean oil system, the oil was the determining co-product. For soybean oil though, it is the demand for the protein meal that determines the supply, not the demand for the oil. Therefore, the modeling of soybean oil did not directly involve the cultivation and processing of soybeans. Instead, a change in demand for soybean oil would affect other users of soybean oil, which would then have to compensate with another oil. The most likely compensation oil was palm oil since this was identified as the marginal one (Schmidt and Weidema 2008). This meant that the environmental impact from demanding soybean oil was similar to that of palm oil. The identification of determining products and byproducts is summarized in Table 1.

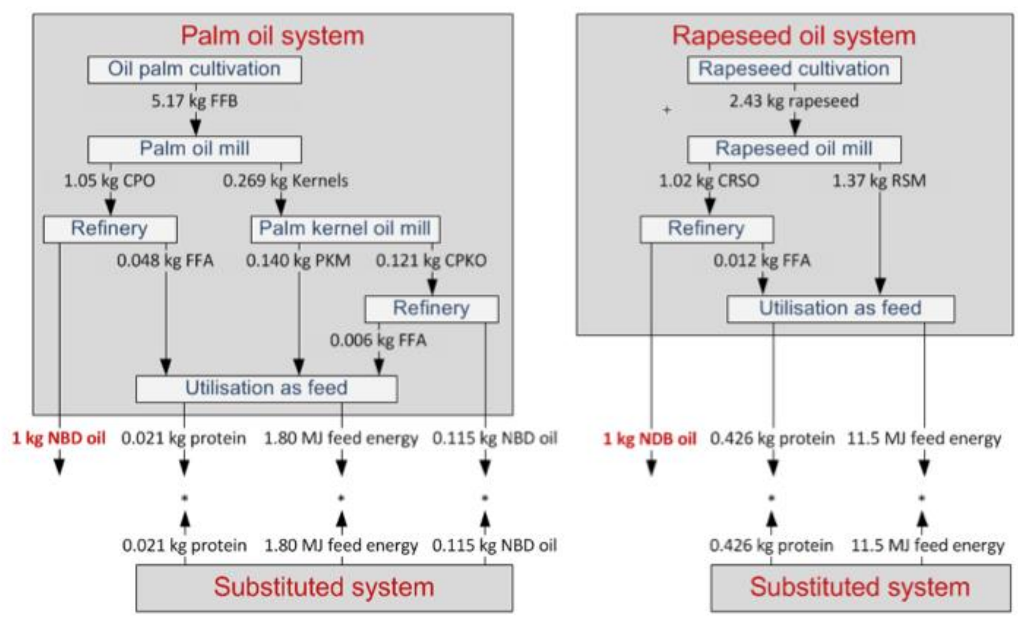

Figure 1. Process diagrams for palm oil (left) and for rapeseed oil as an example of all other oils (right). NBD: neutralized, bleached, deodorized, FFB: fresh fruit bunches, CPO: crude palm oil, CPKO crude palm kernel oil, FFA: free fatty acids, PO: palm oil, PKO: palm kernel oil, PKM: palm kernel meal, RSM: rapeseed meal, CRSO: crude rapeseed oil.

Table 1. Determining products and by-products in the five oil production systems.

	Palm oil system	Soybean oil system	Rapeseed oil system	Sunflower oil system	Peanut oil system
NBD oil	Determining	Byproduct	Determining	Determining	Determining
NBD palm kernel oil	Byproduct	n.a.	n.a.	n.a.	n.a.
Protein meal	Byproduct	Determining	Byproduct	Byproduct	Byproduct
FFA	Byproduct	Byproduct	Byproduct	Byproduct	Byproduct

n.a.: not applicable.

For the substituted systems caused by the by-products, the marginal supply of feed energy and feed protein were identified as barley and soybean meal in Schmidt and Dalgaard (2012, 63-64, 85). Furthermore, Schmidt and Dalgaard (2012), identified the marginal suppliers of barley and soybean meal as Ukraine and Brazil.

The inventoried regions of the five oils were identified as the country/region with the highest increasing trend in production volume of the oil crops from 2001 to 2011 (FAOSTAT 2013a), as this trend indicates which region could be a good candidate for the marginal one, i.e. the country/region that is likely to supply a change in demand for a specific oil. It was assumed that the oil mills and refineries are located in the same region/country as the cultivation of the oil crops. The chosen regions were:

- Fresh fruit bunches: Indonesia/Malaysia
- Soybean: Brazil
- Rapeseed: Europe (EU27)
- Sunflower: Ukraine
- Peanut: India

2.4. Indirect land use change (iLUC)

In the current study an advanced cause-effect based iLUC model described in Schmidt et al. (2012) is applied. This model was developed by 2.-0 LCA consultants through a larger project (2.-0 LCA consultants, 2014) supported by a large range of industries (e.g. Unilever, DuPont, TetraPak, Arla Foods, DONG Energy, United Plantations), universities (e.g. Swedish University of Agriculture Sciences, Aalborg University and Copenhagen University) and other research related organisations (e.g. The Sustainability Consortium, the ecoinvent LCA database, the Roundtable on Sustainable Palm Oil (RSPO) and the Japanese National Agricultural Research Center) plus several others.

This model considers land as capacity for biomass production. There exists a market for land, which is called the land tenure market. Since crops can be grown in different parts of the world and since crops are traded on global markets, it is argued that this market for land is global. The ‘product’ traded on this global market is capacity for biomass production. It should be noted that this capacity can be created in different ways:

1. Expansion of the area of arable land (deforestation)
2. Intensification of land already in use (through e.g. increased use of fertilizers)
3. Crop displacement, i.e. someone reduces consumption, e.g. induced by increases in prices, in order to allow others for using the biomass production capacity (social impacts)

The third point above is assumed to be zero because LCA considers long-term effects of changes in demand. In the long term, suppliers will adjust their production to match demand, and unless the production costs are higher, the prices will remain unchanged.

The capacity for biomass production needs to be measured in an appropriate unit. An obvious option for a reference flow of a land-tenure activity would be occupation of land (ha yr). However, this approach does not take into account that the potential production on 1 ha yr land in e.g. a dry temperate climate is very different from the potential in wet tropical climate. Another option would be the potential Net Primary Production (NPP_0), measured in kg carbon, which was the solution adopted.

As it can be seen in Figure 2, the model accounts for emissions related to both expansion (deforestation) and intensification of land. The proportion between expansion and from intensification is calculated based on the total NPP_0 on new arable land and total NPP_0 (carbon in crops) from an increase in fertilizer use in one year. All inflows to the land-market tenure activity are measured in kg NPP_0 (as kg carbon). The NPP_0 from expansion is determined based on general NPP_0 per ha yr figures (Haberl et al. 2007) and figures on annual increase of arable land. The NPP_0 from intensification is calculated as the carbon in crop produced via intensification during one year. The intensification is determined based on crop yield dose-response figures for fertilizer input (Schmidt 2008) combined with information on which crops and where intensification takes place (data from FAOSTAT) and current fertilizer levels for these crops (IFA 2013).

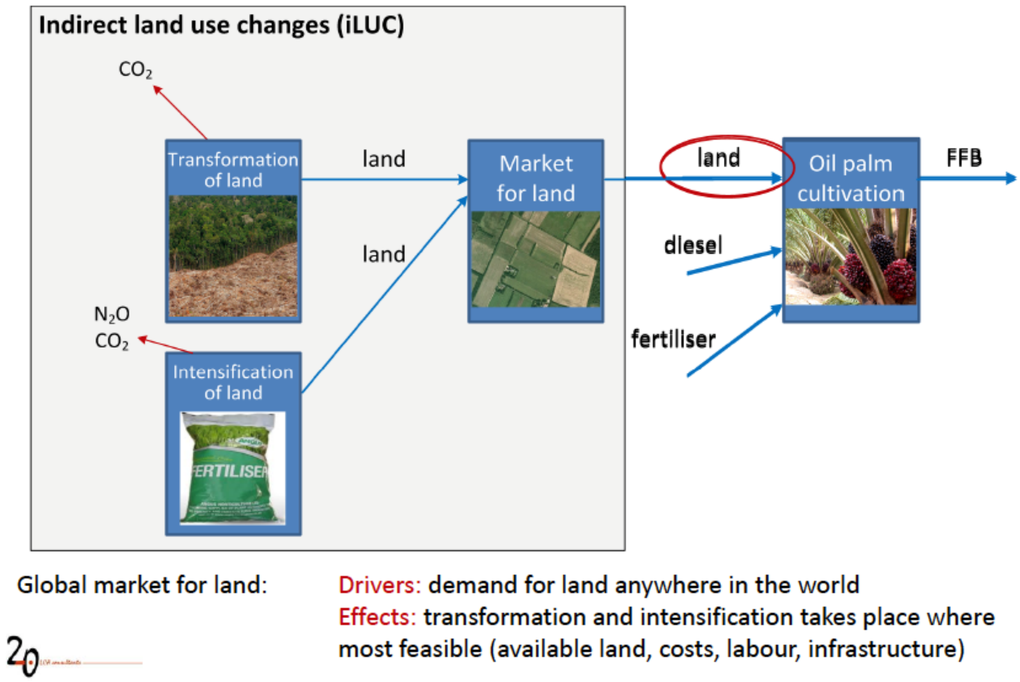

Figure 2. Conceptual representation of the iLUC model used in the study.

When the occupation of land causes deforestation, a critical point is often to decide the period of time over which the deforestation emissions should be allocated or 'amortized'. The current model does not operate with amortization. If only expansion is considered, occupation of 1 ha in 1 year will cause 1 ha deforestation. After the duration of 1 yr, the land is released to the market for land, i.e. to other crops, which can then be grown without deforestation. Hence, the occupation of 1 ha yr is modeled as 1 ha deforestation in year 0 and -1 ha deforestation in year 1. In order to model the GHG effects of this intermediate acceleration of deforestation, the method described in Kløverpris and Mueller (2013) is used.

Overall this iLUC model has several key characteristics that make it superior to many of the other existing models: is applicable to all crops (also forest, range, build etc.) in all regions in the world, it overcomes the allocation/amortization of transformation impacts, and it is based on modeling assumptions that follow cause-effect relationships and standard modeling consistent with any other LCA-processes. It is acknowledged though that this is one among many other models, and that there is currently no consensus in the LCA community on how to model iLUC. Therefore, the contributions to results from iLUC are reported separately.

2.5. Life cycle inventory data sources

The oil crop (and barley) yields were estimated for 2011 by linear regression over the period 2001-2011 with use of data from FAOSTAT. For oil palm a weighted average of Malaysia and Indonesia was applied, based on area cultivated: 40% MY and 60% ID. Fertilizer inputs are identified in the following data sources: Oil palm (Schmidt 2007), Soybean (Dalgaard et al. 2008), rapeseed oil (Plantedirektoratet 2004), Sunflower (FAO 2005), Peanut (Diwakar 2004; Talawar 2004), and Barley (FAO 2005). The fertilizer mixes of different sources of N, P and K are identified at the country level in IFA (2013). Diesel consumption was obtained from Dalgaard et al. (2008), Cederberg et al. (2009) and Schmidt (2007). Water use for irrigation was modeled using data from AQUASTAT (FAOSTAT 2013b) and FAOSTAT (2013a). Oil palms are not irrigated and <1% of the area with barley and sunflower in Ukraine is irrigated. Irrigation data for rapeseed in EU27 were approximated by non-crop specific data for Germany and France. For the iLUC model, the potential productivity of the occupied land was estimated based on Haberl et al. (2007). N-related emissions for all crops (NH_3, NOx, N_2O, NO_3, N_2) were estimated based on a N balance, using the methodology from IPCC (2006).

In average 18% of the cultivated oil palms were assumed to be grown on peat, based on Agus et al. (2013). It should be noticed that this proportion may not reflect the marginal producers of palm oil, but the current average. CO_2 emissions from peat were estimated as 43 t CO_2 per ha yr (for average drainage depth) based on a review of several studies (Agus et al. 2013b; Melling and Henson 2011; Hooijer et al. 2010; Hooijer et al. 2012). N_2O emissions from peat oxidation were calculated by using IPCC (2006).

The inventories for oil extraction in the mills were based on Schmidt (2007), Dalgaard et al. (2008) and Dalgaard and Schmidt (2012) for soybean oil, palm oil, palm kernel oil and rapeseed oil. For sunflower and peanut oils the meal/oil ratios were obtained using statistics on global oil and meal production from the period 00/01 to 05/06 (Oil World annual 2005). The milling operation includes thermal energy inputs, electricity inputs (as well as outputs exported to the grid), transports, water inputs, and capital equipment (mill). The by-products from the milling process e.g. empty fruit bunches, meal, are utilized as fertilizers or feed and thereby substitute mineral fertilizer or feed energy (barley) and feed protein (soybean meal) on the market.

The inventories for the refining process are based on Schmidt (2007) and Schmidt and Dalgaard (2012). These refineries supply refined oils and FFA (free fatty acids). The latter is used for livestock feeding and thereby substitute feed energy on the market.

The ecoinvent database v.2.2 (ecoinvent 2010) was used to model the background system (production of fertilizers, and other materials, energy, and capital goods. This database version is not linked using consequential modeling. However electricity profiles used in the foreground system (Europe, India, Brazil, Indonesia, Malaysia, World) were defined using a consequential approach, as explained in Schmidt et al. (2011).

Inventory data for marginal sources of feed protein (soybean meal from Brazil) and feed energy (barley from Ukraine) were obtained from Dalgaard and Schmidt (2012) and Schmidt and Dalgaard (2012).

2.6. Life cycle impact assessment

Life cycle impact assessment focused on three indicators, namely global warming, land occupation and water consumption. Global warming is calculated using IPCC's GWP100 (Forster et al. 2007). Biogenic CO_2 was included as CO_2 stored in oils. Biogenic CO_2 emissions originating from iLUC were also included. Land occupation is used in this study as an early midpoint indicator for impacts on biodiversity, and it includes only occupation of arable land. With regard to water consumption we used the terminology from the Water Footprint network (Hoekstra et al. 2011), however only 'blue' water (irrigation) is included in the impact assessment, whereas 'green' water (rain water) as well as 'grey' water (pollution) were excluded. Water consumption was assessed with two complementary approaches, namely consumption in volume (m^3), and consumption after characterization using the Water Stress Index (WSI) from Pfister et al. (2009), measured in m^3-eq.

3. Results and discussion

3.1. Results per ton of oil

Figure 3 shows the results of the impact assessment. The results for soybean oil and palm oil are equal. This is explained by the fact that soybean oil is a dependent co-product, thus an extra demand for soybean oil does not affect the soybean oil production, but instead the production of palm oil. The GHG emissions include in all oils a credit due to the embedded carbon in the oils, of 2.8 tons CO_2 per ton oil. It is common in many studies to omit this short-term storage of carbon in products, given that it is expected to be compensated by an equal emission at the end of life. In order to compare our results with studies using this approach, the GWP results in Figure 1 would need to be changed by means of adding 2.8 tons CO_2 per ton oil, for all oils.

When iLUC emissions are considered, the overall GHG emissions are considerably higher for peanut oil compared to the others. This is mainly because the land occupation per ton refined oil is high (as shown in the land occupation graph in Figure 1). High land occupation results in high iLUC-related GHG emissions. When iLUC emissions are excluded, the overall GHG emissions are reduced for all oils, and the ranking changes slightly, with sunflower oil as the best performer, whereas when iLUC is included rapeseed oil is the best performer. For the other oils, iLUC does not affect the ranking.

The results for blue water use show that palm oil, soybean oil and peanut oil have a net consumption of water while rapeseed oil and sunflower oil are associated with a net water saving. This may seem surprising because both rapeseed and sunflower are irrigated. However, the by-products of rapeseed oil and sunflower oil, i.e. the oil meals, displace soybean meal, barley, and palm oil. Since soybeans and barley are irrigated more than rapeseed and sunflower, the net use of water becomes negative. Palm oil is associated with a low water use because it is not irrigated, however it is not zero due to contributions from the oil mill, refineries, production of chemicals, fertilizers, machinery, etc. When the WSI is applied, it does not change the ranking of oils as compared to blue water by volume.

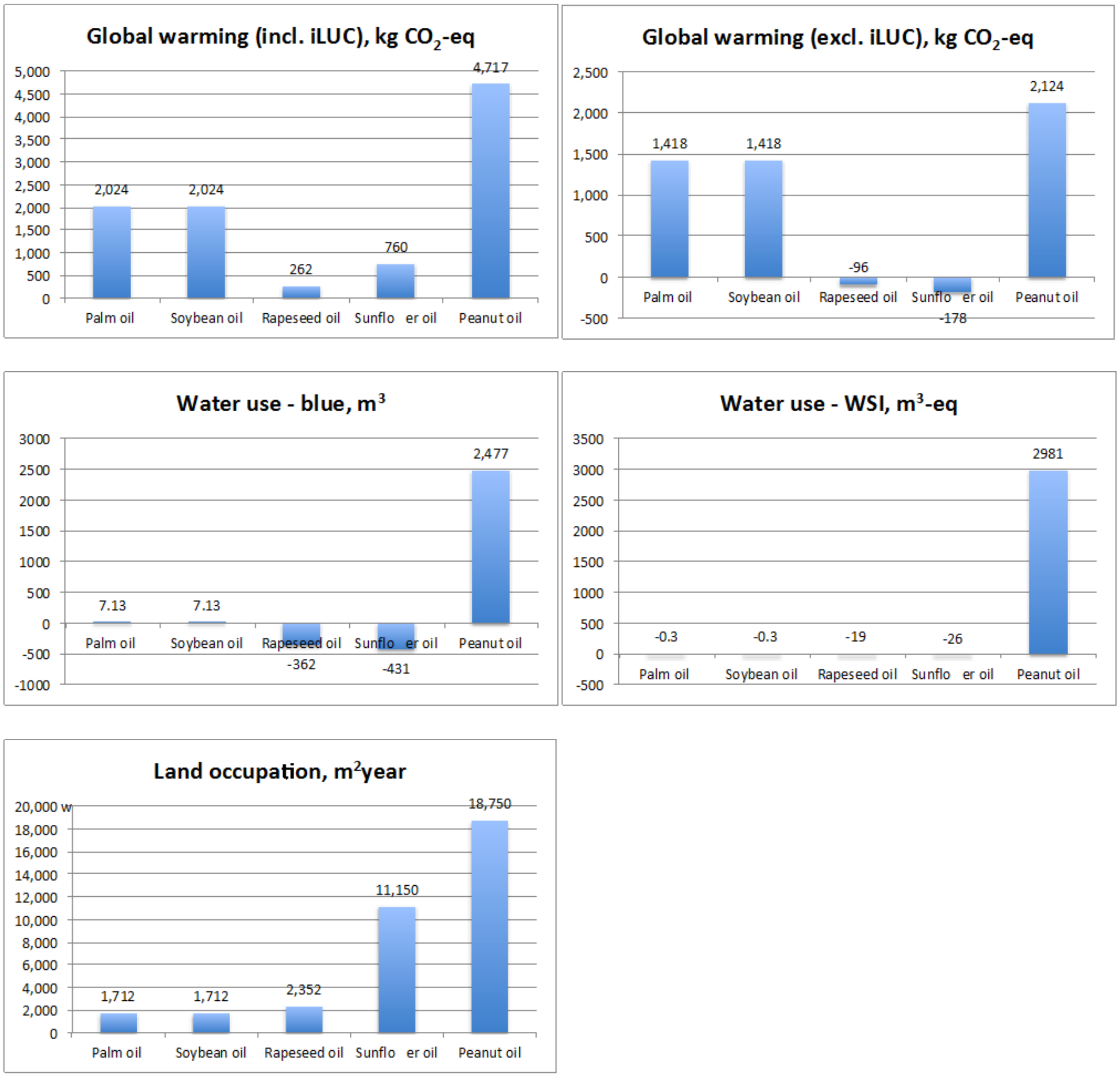

Figure 3. Results for five different refined vegetable oils, per ton refined oil. Global Warming includes biogenic carbon sequestration in the oil.

With regard to land occupation, the highest contribution for all oils is caused by the crop from which each of the oils is extracted. Other contributions are related to substitutions caused by by-products. Even accounting for substitutions, it is the land occupied by the oil crop itself the one that determines the main impact. The best-performing oils are palm oil and soybean oil, followed by rapeseed oil. Sunflower oil and peanut oil, in this order, are the ones with the highest land occupation.

3.2. Sensitivity analyses focused on GHG emissions

A number of sensitivity analyses were carried out, with a focus on GHG emissions. They are summarized in Table 2.

Table 2. Sensitivity analysis summary.

Parameter	Description	Sensitivity analysis outcome
Palm oil: FFB yields	Oil palm is cultivated with Malaysian and Indonesian yields respectively, rather than weighted average.	Low sensitivity, ranking unchanged
Palm oil: Share of oil palm area on peat soils	Default share of peat (18%) compared to cultivation on mineral soil (0% peat), 11% peat (Malaysian average) and 22% (Indonesian average).	Low peat share leads to palm oil with similar GHG emissions to rapeseed oil
Palm oil: CO_2 emission factor for peat soils	Default emission factor of 43 ton CO_2/ha yr compared to 27.5 and 85.5 ton CO_2/ha yr	Sensitive but ranking unchanged
All oils: Fertilizer manufacturing (nitrous oxide emissions from nitric acid)	Default emission factor is 8.39 g N_2O/kg HNO_3 compared with 4.5 g N_2O/kg HNO_3 (IPCC default value) and 1 g N_2O/kg HNO_3 (representing best available technique).	Low sensitivity, ranking unchanged
All oils: Source of marginal barley (feed energy)	Marginal supplier of barley (Ukraine) changed to Argentina.	Low sensitivity, ranking unchanged
All oils: iLUC model	Default iLUC proportion between transformation and intensification is 37% and 63%, respectively. This is changed to transformation and intensification being 90% and 10%, respectively.	Low sensitivity, ranking unchanged

The most significant individual sources of uncertainty related to GHG emissions were identified as the share of oil palm grown on peat, and the CO_2 emission factors for peat soils. If the share of peat is very small, the GHG emissions for palm oil become close to rapeseed oil. In the future, when the palm oil expansion may take place in other regions than Indonesia and Malaysia, the issue of peat may change if this will take place in e.g. Africa or Latin America.

3.3. Uncertainties in water use

The data used for water consumption are often not crop-specific. Hence, these data are associated with significant uncertainties. Substantial uncertainty is also associated with the determination of WSI values, given that the exact location where crops are grown is not known in detail, and average values for regions, countries or groups of countries need to be used.

4. Conclusion

A consequential cradle-to-gate LCA has been carried out on five major vegetable oils. The environmental impacts assessed were GHG emissions, water consumption and land occupation. When we look at the environmental impacts of demanding an additional ton of oils, we see clear tradeoffs. Rapeseed oil and sunflower oil perform best in GHG emissions, sunflower oil in water consumption and palm oil and soybean oil in land use. The poor performance of peanut oil stands out in all impact indicators.

It was found that iLUC emissions have a substantial contribution to GHG emissions, especially in those oils with relatively low yields. As for water consumption, the use of WSI factors didn't change the ranking of oils, but mainly increased the differences between them.

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Social indicators for meat production – addressing workers, local communities, consumers and animals

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ABSTRACT

Social impacts regarding the production of meat are on the agenda, especially due to reoccurring problems, like food hygiene, the use of antibiotics and disregard of animal's welfare. Thus, the assessment of social impacts for meat production seems essential, but has not been determined within social life cycle assessment (SLCA), yet. As pork has the biggest proportion of the global meat production per mass; the focus of this study is on pork production. The SLCA method is used to address the related impacts. Currently it addresses only human beings, while completely neglecting the impacts on animal's life. Therefore, this study aims to integrate animals into SLCA alongside the common stakeholder groups' workers, local communities, and consumers. Indicator sets are developed to assess the value chain of pork. All indicators are allocated to relevant midpoint impacts to indicate impact pathways. Further research is needed regarding the characterization of occurring impacts.

Keywords: social life cycle assessment, animal well-being, indicator development, meat production

1. Introduction

Social impacts regarding the production and consumption of meat are on the agenda. Meat consumption is increasing worldwide. This increase in consumption originated in western civilization, but is also arising in newly industrialized countries like the BRICS[1] states. This causes several problems for societies on a global scale. Most of the occurring problems are connected to industrial animal farming. Feedstuff production is mostly outsourced resulting in monocultures. Animal farming, which is detached from feedstuff production, creates great quantities of manure. Relevant topics in connection with the meat supply chain are popping up not only in the media but also in politics and public discussions. It starts with working conditions of farmers for both animal feedstuff and animal husbandry. Further topics of interest are working conditions of abattoir workers, food hygiene, the usage of antibiotics and hormones within animal farming, and last but not least the animal treatment primarily within industrial farms (Safran Foer 2010; Chemnitz et al. 2014). Especially the latter is gaining increasing awareness within different movements of society. Although, the status of animal rights has not been fully clarified, interest in animal's well-being in connection is increasing, as current practice contradicts with common ethical values of society (Sunstein and Nussbaum 2004; Singer 2006; Daigle 2014). However, current assessment practice of food products in general including meat products is limited to life cycle assessment (Dalgaard 2007; Roy et al. 2009), but no social life cycle assessment (SLCA) has been performed addressing meat production chains. No set of indicators exists considering affected stakeholders due to animal husbandry, feedstuff and meat production or meat consumption processes. Hardly any impacts along the supply chain are covered by current methods. In addition, common SLCA practice (Benoit and Mazijn 2009) is completely neglecting the impacts on animal's life. Facing these shortcomings within this study a set of indicators is developed addressing workers, local communities, consumers and animals. It is focused on pork meat and the related production chain, as it has the highest market share per mass and records increasing rates for emerging countries (Chemnitz et al. 2014).

2. Methods

The pork production chain is examined focusing on the inclusion of indicators by considering feedstuff production, animal farming and slaughter and the consumption of the final product pork (see Figure 1). The focus is on intensive livestock farming, as it contributes with the highest amount to the pork supply. In this connection global supply chains are relevant, as e.g. feedstuff production is up to a great extend outsourced (BLE 2011).

[1] BRICS states are Brazil, Russia, India, China and South Africa.

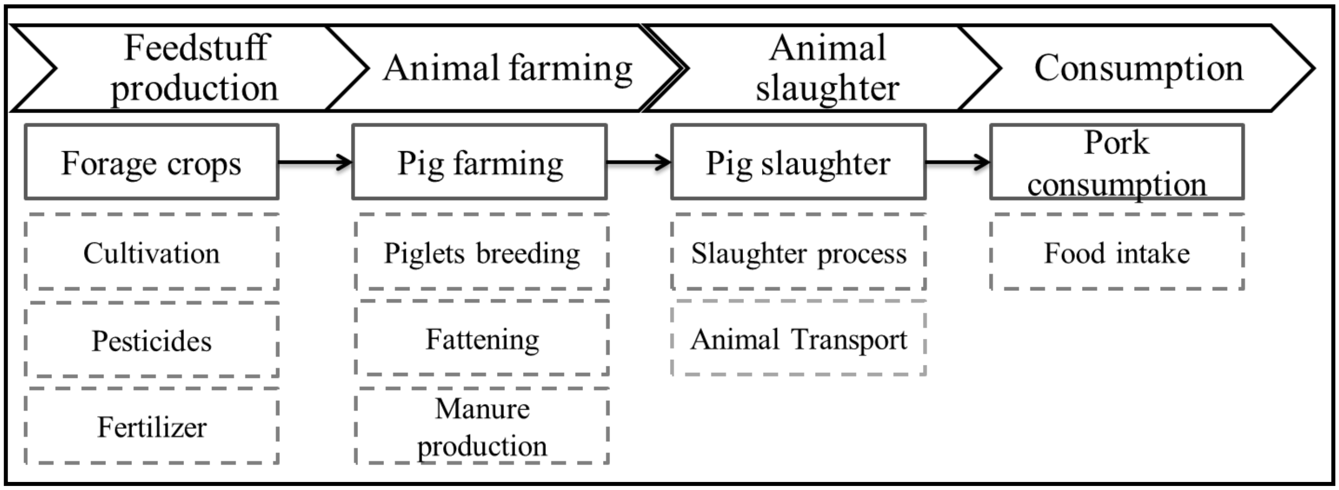

Figure 1: Pork production chain including feedstuff production, animal farming, animal slaughter and the pork consumption; all relevant steps from a SLCA point of view are included

The SLCA method is used, targeting the development of a set of indicators that integrates human as well as animal well-being, which so far is not included into SLCA. It is orientated on the Guidelines of SLCA (Benoit and Mazijn 2009) and the related Methodological sheets (Benoît et al. 2013). It is focused on indicators in connection with midpoint impact categories, addressing aggregated repercussions on human and animal well-being. In addition, existing indicators are analyzed, which are not yet included into SLCA practice. Subsequently, additional indicators are developed to better match the broad range of appearing impacts. Indicators used in SLCA studies are qualitative, semi-quantitative or quantitative (Dreyer et al. 2006) and are provided by different organizations. The Methodological sheets provide guidance for case studies based on indicator approaches, exemplary data sources and relevant international conventions (e.g. conventions of the International Labour Organization) (Benoît et al. 2013). In order to enable a preferably broad indicator review the Global Reporting Initiative (Global Reporting Initiative 2013) and the Product Sustainability Assessment Guidance published by the Institute of Applied Ecology (Grießhammer et al. 2007) are taken into account, too.

According to the guidelines of SLCA (Benoit and Mazijn 2009) five stakeholder groups are defined: workers, local communities, societies, consumers and value chain actors. The stakeholder groups affected can be different dependent on the respective case study. Commonly addressed is the stakeholder group workers, which is in close relation to production processes (Norris 2006b; Klöpffer 2008; Jørgensen 2012). Local communities and consumers are also often considered, e.g. in connection with health aspects. Health issues even if originally located in LCA are shifted to SLCA to avoid double counting and to ensure a consistent separation of social, environmental and economic issues. Value chain actors and societies are more relevant at the organizational perspective, reflecting social responsibility activities (Benoît et al. 2013). Animals are thus far not addressed within SLCA.

Within SLCA two topics are always addressed: workers or working conditions and health aspects (Hunkeler 2006; Norris 2006a; Weidema 2006; Parent et al. 2010; Jørgensen 2012). Besides that another important topic, which is listed quite frequently as well is education. According to the Methodological sheets six midpoint impact categories are defined: human rights, working condition, health and safety, cultural heritage, governance and socio-economic repercussions. However, Benoît et al. (2013) states that these categories can be complemented by further ones. Therefore fair wage and level of education are suggested as additional midpoint categories by (Neugebauer et al. 2014).

3. Considering animals in SLCA

So far animals are not considered within SLCA. However, the consideration of animals or animals´ well-being in connection with animal husbandry and farming already has a great history in philosophy. Several philosophers have already discussed the topic of animal rights and the moral responsibility of mankind´s behavior with respect to animals. Generally it is focused on the differences and similarities between species and the resulting rights for each species. Highlighted topics in this context are the ability of most animals to feel not only pain and pleasure but also the existence of the inherent will to survive and the presence of future plans (Regan 2004;

Sunstein and Nussbaum 2004; Singer 2006). Especially in connection with industrial animal farming[2] these topics become highly relevant. Already in the early sixties Harrison (1964) stated the situation of poultry in industrial farms, which resulted in the first public awareness actions against industrial farming. It served as a kick-off for the Brambell Report in 1965 (Brambell 1965) including the five freedoms for farm animals[3]:

1. Freedom from hunger and thirst - by ready access to fresh water and a diet to maintain full health and vigor
2. Freedom from discomfort - by providing an appropriate environment including shelter and a comfortable resting area
3. Freedom from pain, injury or disease - by prevention or rapid diagnosis and treatment.
4. Freedom to express normal behavior - by providing sufficient space, proper facilities and company of the animal's own kind.
5. Freedom from fear and distress - by ensuring conditions and treatment which avoid mental suffering.

The five freedoms serve as the basis of the European Convention for the protection of animals kept for farming purposes (Council of Europe 1976) and have also been included in the endeavors of the Welfare quality network[4]. However, even if several actions exist the reality shows different, when taking a closer look at the everyday practice in industrial farms. Almost constantly the points 3)-5) are harmed (Safran Foer 2010; FAWC 2011; Chemnitz et al. 2014). Therefore, there is a need to include animal’s welfare or well-being into SLCA to address the impacts on animal’s life.

4. Results

According to the above mentioned, indicators are selected, developed and allocated to the respective stakeholder group and the affected midpoint category. For each step of the defined life cycle (see Figure 1) relevant stakeholder groups are identified and the related effects are analyzed. Thereafter relevant indicators are taken from existing SLCA studies complemented by own indicators developed to tackle all analyzed effects.

Within the defined life cycle the two stakeholder groups’ workers and local communities are directly affected. Workers might be impacted in all production processes from feedstuff production to the animal slaughter, as none of the related processes can get along without manual work. Local communities are always influenced, when boundaries between agriculture and living areas overlap and confounders result from the agricultural activities. The stakeholder group consumers are affected at the end of the supply chain, due to the consumption of pork. The remaining stakeholder groups, value chain actors and societies, are not included in this study, as they are mainly related to companies’ behavior reflecting organizations understanding of promoting social responsibility (Benoît et al. 2013). The named stakeholder groups reflect the interest of human well-being, but neglect the impacts on animal´s life resulting from animal farming. Addressing these shortcomings, indicators are included representing the impacts on animals life. Following the defined midpoint impact categories for humans (see section 2) three midpoint impacts are considered: animal rights, animal health and species specific repercussions. As the status of animal rights is not finally clarified, it is seen as a subordinated midpoint category. The remaining categories (e.g. fair wage) are assumed as irrelevant for the well-being of animals. The discussion of how to include the proposed indicators into the existing SLCA framework cannot be solved easily, but proposals are given in section 4.4.

4.1. Stakeholder group workers

Along the supply chain for pork production different groups of workers are affected. According to Figure 1 workers in feedstuff production, animal farming and slaughterhouses are considered. Within the feedstuff production the focus is on cereals and soy production processes. Whereas cereals are often produced locally soy production happens mainly in South America (Burley 2008). Workers are impacted due to land grabbing (Burley

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[2] Of course also in connection with animal testing and animal husbandry, but within this study the focus is clearly on animal farming.
[3] Provided by the Farm Animal Welfare Council (http://www.fawc.org.uk/freedoms.htm)
[4] Can be accessed under http://www.welfarequalitynetwork.net/network

2008; Fritz 2011), forced labor (Sakamoto 2010), disrespect of indigenous rights (Greenpeace 2006; Fritz 2011) and health impacts caused by pesticides (PAN 2010; Fritz 2011).

For workers within the industrial pig farms different impacts are anticipated. The standard of living in connection with indebtedness and fair wages seems to be important, as the historical development of pig farming in Germany shows trends towards mass based industrial farming achieving a minimum of economic efficiency (Korbun et al. 2004). In addition health impacts are relevant due to the existence of MRSA[5] germs on pig farms (Neeling et al. 2007; Khanna et al. 2008; Frick 2010) and frequent cases of chronic bronchitis and asthma of farmers (Iversen et al. 1988; Vogelzang et al. 1999; Iversen et al. 2000; Radon et al. 2001). Besides, as most of the reviewed farms within this study[6] are family businesses, child labor might have a small impact. The respective regulations and laws therefore have to be considered.

Workers within modern slaughterhouses are forced to perform a high number of slaughters per hour to be cost-efficient. Thus, the workers are exposed to high psychological pressure, which may result in abnormal behavior (Bundestierärztekammer 2010; Safran Foer 2010; Albert Schweitzer Stiftung 2011).

According to these findings in Table 1 a list of indicators is presented. Indicators are taken from the Methodological sheets, the GRI criteria and the PROSA report complemented by the authors of this study´s own indicators. Relevant midpoint impacts are allocated to the indicators based on the definitions in section 2.

4.2. Stakeholder group local community

Local communities are affected in connection with the feedstuff production but also by pig farming and slaughter. The already mentioned soy production in South America is heavily impacting local communities through land use, relocations, land grabbing and health effects caused by pesticides (UNFPA 2001; Casson 2003; PAN 2010; Fritz 2011). In addition, feedstuff production is affecting local communities through the pollution of groundwater (Anderson and Sobsey 2006). Complaints partly are noted regarding odor caused by manure used for fertilizing.

The pig farming itself impacts local communities by noise and odor (e.g. through ammonia emissions) and the potential loss of work places, which causes at least in Germany, waves of protest against industrial farm projects (BUND 2013). Additional factors are health effects, caused by emissions (e.g. ammonia, antibiotics, sulfurous substances and bacteria) from accruing manure (Wing and Wolf 2000; Nimmermark 2004; Walker et al. 2007; European Commission 2011).

Problems occurring in the neighborhood of slaughterhouses[7] are high traffic volumes and related noise impacts, but also groundwater pollution with pathogenic agents, fats and excrements (Johns 1996; BMLF 1999).

Indicators addressing these impacts can be taken from Table 2. They are connected to relevant midpoint impacts. Indicators are taken from the Methodological sheets complemented by the authors´ own developments.

[5] Methicillin-resistant Staphylococcus aureus
[6] As part of the study different pig farmers in Germany have been interviewed.
[7] Nowadays slaughterhouses are mostly highly efficient and highly industrialized operating units with a high number of animals killed per hour.

Table 1: Indicator selection for the stakeholder group workers divided in general, feedstuff production, pig farming and pig slaughter indicators, complemented by new indicators and allocated to relevant midpoint impacts

Worker		
	Indicators	**Impacts[1]**
General indicators	Existence of labor laws per organization, sector and country [a]	WC, H&S, FW
	Potential of country/organization not passing labor laws [b]	WC, H&S, FW
	Violations of obligations to workers under labor or social security laws and employment regulations [b]	HR, WC, H&S, FW
	Freedom to join unions and to perform collective bargaining [b]	HR, WC
	Existence of transparent wage systems [c]	WC, FW
	Highest/lowest wage paid per organization, sector and country [b]	WC, FW
	Minimum and non-poverty wages per organization, sector and country [b]	WC, FW
	Duration and way of continued pay in case of sickness [c]	WC, FW
	Existence of contracts for work and labor [a]	WC
	Existence of regulations of working hours and overtime arrangements [b]	WC
	Compliance of legal recovery times [a]	WC
	Cases of discrimination and actions taken per organization, sector and country [d]	HR, WC, SER
	Ratio of basic salary of men to women by employee category [d]	HR, WC, FW, SER
	Accidents per organization, sector and country [b]	H&S
	Injuries, occupational diseases and lost days per organization, sector and country [d]	H&S
	Social benefits provided to the workers (e.g. health insurance, pension fund, child care, education, etc.) [b]	WC
	Existence and cases of child labor per organization, sector and country [b,c]	HR, WC, SER, E
	Existence and cases of forced labor per organization, sector and country [b,c]	HR, WC, SER, E
Feedstuff production	Health effects due to pesticide use per organization, sector and country [a]	H&S
	Prevalence of racial discrimination per organization, sector and country [b]	HR, SER
	Indigenous land rights conflicts/land claims [a,b]	HR, SER
Pig farming	Indebtedness per organization, sector and country [a]	WC, SER
	Excessive working hours per organization, sector and country [b]	WC, H&S
	Occurrence of chronicle diseases per organization, sector and country [a]	H&S
	Health effects due to (resistant) germs per organization, sector and country [a]	H&S
Pig slaughter	Number of slaughtered animals per hour and worker per organization, sector and country [a]	WC
	Psychological health effects per organization, sector and country [a]	H&S

[1] Midpoint impact categories allocated to the defined indicators according to Benoît et al. (2013) and Neugebauer et al. (2014); see section 2 – found to be relevant for workers are human rights (HR), working condition (WD), health & safety (H&S), social-economic repercussions (SER), level of education (E) and fair wage (FW)
[a] Developed indicators by the authors of this study
[b] Indicators taken from the Methodological sheets of SLCA (Benoît et al. 2013)
[c] Indicators taken from the Product sustainability assessment guidance (Grießhammer et al. 2007)
[d] Indicators taken from the Global Reporting Initiative (Global Reporting Initiative 2013)

Table 2: Indicator selection for the stakeholder group local community divided in general, feedstuff production, pig farming and pig slaughter indicators, complemented by new indicators and allocated to relevant midpoint impacts

Local community		
Indicators		Impacts[1]
General indicators	Changes in land ownership [b]	HR, SER
	Use of abiotic and biotic resources and water per sector, area and country [a,b]	SER
	Prevalence of conflicts based on resource use per sector, area and country [b]	SER, H&S
	Freedom of expression and protest per sector, area and country [b]	HR, SER
	Relocations of persons per sector, area and country [b]	HR, SER
	Protection of cultural heritage per sector, area and country [b]	SER
	Health effects and diseases per sector, area and country [b]	H&S
Feedstuff production	Health effects due to pesticide use per sector, area and country [a]	H&S
	Pollution of water and groundwater due to pesticides per sector, area and country [a]	H&S, SER
	Prevalence of racial discrimination per sector, area and country [b]	HR, SER
	Indigenous land rights conflicts/land claims per sector, area and country [a,b]	HR, SER
Pig farming	Pollution of water and groundwater due to (resistant) germs per sector, area and country [a]	H&S, SER
	Occurrence of chronicle health effects per sector, area and country [a]	H&S, SER
	Increase in noise and odor per sector, area and country [a]	SER, H&S
	Health effects due to (resistant) germs per sector, area and country [a]	H&S, SER
Pig slaughter	Pollution of water and ground water per sector, area and country [a]	HR, SER, H&S
	Increase in traffic, noise and odor per sector, area and country [a]	SER, H&S

[1] Midpoint impact categories allocated to the defined indicators according to Benoît et al. (2013) and Neugebauer et al. (2014); see section 2 – found to be relevant are human rights (HR), health & safety (H&S) and social-economic repercussions (SER)
[a] Developed indicators by the authors of this study
[b] Indicators taken from the Methodological sheets of SLCA (Benoît et al. 2013)

4.3. Stakeholder group consumers

Consumers are affected by the final product pork, which per se does not cause negative effects. However, health effects are becoming more relevant, when considering the societal background and the overconsumption of meat in western societies. This overconsumption correlates with common civilization diseases like cardiovascular diseases, diabetes mellitus and some cancers (Walker et al. 2007; McMichael et al. 2007). Further, risks have been determined in connection with the use of antibiotics in animal farming and resulting resistances of bacteria and health effects on the consumer side (Böckel 2008; European Commission 2011; BfR 2014). Open questions remain in connection with genetically modified feedstuff (e.g. soy beans) and possible health effects. With this regard, information provided about the product by the producer to the consumer is clearly lacking. More research is needed, as no valid data are available, yet.

Relevant indicators can be taken from Table 3. Indicator sources are the Methodological sheets and the PRO-SA report complemented by own developments. The indicators are allocated to relevant midpoint impacts.

Table 3: Indicator selection for the stakeholder group consumer divided in general and pork consumption indicators, according to the analyzed effects, complemented by new indicators and allocated to relevant midpoint impacts defined

Consumer		
Indicators		**Impacts[1]**
General indicators	Complaints of consumers per organization, sector and country [b]	SER
	Amount of information on/about the product [b]	SER, H&S
	Presence of law/norm regarding product´s transparency per organization, sector and country [b]	SER, H&S
	Results from hygiene and quality checks per product, organization, sector and country [a]	H&S
	Health effects and diseases resulting from the products use [a,c]	H&S
Pork consumption	Labelling and transparency regarding genetically modified products within the supply chain [a]	(HR), H&S, SER
	Information about a balanced diet per sector and country [a]	H&S, E
	Occurrence of civilization diseases resulting from products use [a]	H&S
	Health effects due to (resistant) germs resulting from products use [a]	H&S

[1] Midpoint impact categories allocated to the defined indicators according to Benoît et al. (2013) and Neugebauer et al. (2014); see section 2 – found to be relevant are human rights (HR), health & safety (H&S), social-economic repercussions (SER) and level of education (E)
[a] Developed indicators by the authors of this study
[b] Indicators taken from the Methodological sheets of SLCA (Benoît et al. 2013)
[c] Indicators taken from the Product sustainability assessment guidance (Grießhammer et al. 2007)

4.4. Animals and assignment within SLCA

Animals are impacted during the processes of pig farming and the slaughter at the end of the value chain. During the farming process the pigs are constantly restricted in their natural behavior[8], due to limited space (e.g. limitations in lying down, in sexual behavior or piglets nursing) and abilities in connection with livestock density (e.g. limitations for cooling, like mud bathing). Further various psychological and physical health impacts are often observed, e.g. digestive diseases, skin and claw injuries and behavioral disorders, expressed in aggressions against other animals (LAVES 2010; Rutherford et al. 2011; Albert Schweitzer Stiftung 2013). Therefore the curly tails and canine teeth are cut, which mostly happens without any anesthesia. Existing laws for animal protection do not see any contradiction with common animal welfare principles, as long as it happens before a certain age of the piglets (BMJV 2013). The same accounts for the castration of male piglets. In addition, due to the high number of animals in a limited space the risk of epidemics is increased (BMJV 2013).

Further, animals are heavily affected by the transportation to the slaughterhouses and the pre-slaughter stopover. The fact that the animals will not survive this step in the product life cycle is neglected within this study, as the argumentation in favor of ending the animals life cannot be solved within SLCA and neither is there any consensus about a justification or right to kill animals for food reasons. However, the transportation of the animals involves a high level of stress often resulting in circulatory collapse or death. During the slaughter process failures in anesthesia occur on a regular basis. Further the broadly used CO2 anesthesia causes additional stress through defensive reaction of suffocation (Provieh 2003; Wendt 2006; Puttrich 2012).

Indicators reflecting these impacts are included in Table 4. Included midpoint impacts are animal rights, which are seen as subordinated (see section 4), animal health and species specific repercussions.

[8] Pigs have a distinctive social behavior, which is described in detail in (Hörning et al. 1999).

Table 4: Indicator selection for the stakeholder group animals divided in general, pig farming and pig slaughter indicators, according to the analyzed effects, allocated to relevant midpoint impacts defined

Animals		
	Indicators [a]	**Impacts[1]**
General indicators	Restrictions in species specific behavior (e.g. reproductive behavior, offspring nursing, social behavior etc.) per organization, sector and country	AR, AH, SSR
	Injuries and occupational diseases per organization, sector and country	AH
	Behavioral disorders per organization, sector and country	AH
	Health effects due to (resistant) germs per organization, sector and country	AH
	Mutilations performed per organization, sector and country	AR, AH
	Fattening period compared to natural live expectation	AR
	Genetic descent per organization, sector and country	SSR
	Freedom of hunger, thirst and pain	AR, AH
	Quality, dimension and hygiene of stables per organization, sector and country	AH
	Livestock density per organization, sector and country	AR, AH
Pig farming	Occurrence of death, injuries and behavioral disorders due to farming practice per organization, sector and country	AR, AH
	Distance and means of transport of piglets to the fattening farm per organization, sector and country	AH
Pig slaughter	Distance and means of transport to the slaughterhouse per organization, sector and country	AH
	Occurrence of injuries, fear and failure in anesthesia during the slaughter process	AH
	Guarantee of a painless and quick death	AR

[1] Midpoint impact categories allocated to the defined indicators; inspired by the already defined midpoint impacts according to Benoît et al. (2013) the following midpoint impacts for animals are defined: animal rights (AR), animal health (AH) and species specific repercussions (SSR)
[a] All indicators are developed by the authors of this study.

The assignment of animals within SLCA is not straightforward, as no stakeholder group exists to represent animals´ well-being. The Guidelines of SLCA define stakeholder groups as “a cluster of stakeholders that are expected to have shared interests due to their similar relationship to the investigated product systems” (Benoit and Mazijn 2009). According to Freeman (2010) a stakeholder is any group or individual who can affect or is affected by a certain action. Even if both sources originally focused on human beings, animals could be defined as a group of individuals affected by livestock farming processes, which shared interests due to their similar relationship to the product. However, the inclusion of animals within an own stakeholder group may cause inconsistencies with existing stakeholder groups, as e.g. children are also not defined as an own group, but as a subcategory. Further, animals cannot advocate for their own concerns and thus would need representatives (e.g. NGOs) to fight for their rights. The inclusion in existing stakeholder groups causes some shortcomings and difficulties as well. However, two groups seem reasonable at first sight: consumers and workers. By including animals into the group consumers, the animals´ interest is only represented through the consumers´ judgment or value choice, contrary to e.g. child labor which is also represented in the stakeholder group workers. Thus, the risk of not representing animals´ interest adequately is quite high. When including animals as a subcategory into the stakeholder group workers, problems occur, as animals are not workers per definition. Since the stakeholder group society only covers societal impacts of organizations, animals cannot be represented within this group, even though industrial farming could indirectly harm the ethical values of society. As a consequence, further research is needed to define a proper place for the inclusion of animals’ well-being within the stakeholder concept of SLCA. Until this issue is resolved, the proposed indicators should be used as an additional set of indicators independent from a particular stakeholder group.

5. Discussion

The presented indicator sets for animals and the three stakeholder groups reflect the life cycle of pork production. However, the indicators listed do not raise a claim of completeness, but rather address common hotspots within the supply chain. The same accounts for the considered midpoint impacts. In addition, impact pathways including characterization factors are not part of this study.

For the first time animals are included into SLCA, but animal rights and the treatment of animals is still under discussion. Thus, one might say that animals cannot be equated with the defined stakeholder groups. The proposal within this study is rather targeting the general inclusion of animals into SLCA, than to start discussions about the equality between different stakeholder groups or subcategories. However, more research is needed to solve the assignment problem of animals in connection with the existing SLCA framework.

Further, some stakeholder groups are not considered in SLCA in general or within this study, e.g. future generations. According to Klöpffer (2008) the inclusion of future generations is of importance for sustainability assessment. Agricultural activities in relation with animal farming are not unlikely to affect future generations by overusing abiotic resources like soil and water. Therefore it is recommended to address future generations within SLCA in foreseeable time.

6. Conclusion

The proposed set of indicators can serve as a starting point for SLCA studies of meat production and the inclusion of animal well-being in current assessment practice. Based on the indicator selection in connection with the affected midpoint impacts the logical next step would be the practical implementation into SLCA case studies. Adaptations are probably necessary and further indicators are possibly needed. However, the provided set of indicators serves as a first step towards the implementation of social aspects for meat production chains especially for the production of pork into SLCA. In addition for the first time impacts on animal's life are considered within SLCA.

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Utility of spatially explicit LCA for agricultural territories

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ABSTRACT

Life Cycle Assessment (LCA) is a methodological framework that estimates environmental impacts of products, systems or services in a life cycle perspective at local to global scales. Some environmental impacts, however, can vary depending on the characteristics of their surroundings and therefore on the location of the activity. This variability can be taken into account in impact assessment using spatialized LCA. Spatial differentiation is especially relevant when studying territories (ca. 100-10,000 km^2), which often have high heterogeneity in environmental characteristics. In this study, we focus on defining a method to apply spatialized LCA to study land-use planning in an agricultural territory. In light of agricultural impact assessment, we suggest a future method for spatialized territorial LCA, which is currently being tested on an agricultural territory, the *Lieue de Grève* watershed in Brittany, France.

Keywords: life cycle assessment, spatialization, territory, land-planning, agriculture

1. Introduction

Currently, most agricultural and urban land-planning decisions are made at the territorial level. Since rural and urban development is increasingly addressed at the territorial level, developing a method to estimate environmental impacts of a territory seems important. The concept of "territory" is still debated and varies among and within scientific communities and countries. We focus on agricultural territories, defined as geographically delimited areas in which the majority of land use is based on agriculture. Agriculture contributes to a range of impacts on the environment, and some of these impacts vary depending on the biophysical context in which potential pollutants are emitted (climate, soil type, slope, hedgerow presence …).

In Life Cycle Assessment (LCA), potential environmental impacts are calculated through all stages of a product's or system's life (from cradle to grave). Many LCA studies of agricultural production focus on the field or farm level (Brentrup et al. 2004a,b; Nguyen et al. 2013; Prudêncio da Silva et al. 2014; van der Werf et al. 2009). Focusing on the territorial level, however, allows interactions between farms to be assessed.

Loiseau et al. (2012) highlighted the use of LCA to estimate potential environmental impacts of a territory for land-use planning purposes, such as where to locate different activities in a territory to reduce their environmental impacts. Due to the large size of territories (100-10,000 km^2), the use of spatialized data seems necessary to estimate environmental impacts accurately. This is especially relevant for an agricultural territory, where many emissions and impacts depend on their surroundings. It thus seems relevant to develop a method to estimate environmental impacts of activities within an agricultural territory ("territorial LCA") by considering their locations in a spatially explicit manner. The method developed can be called "spatialized territorial LCA".

We propose to apply the principles of spatialized territorial LCA to determine how and why it can support territorial approaches for assessing environmental impacts of an agricultural territory. The method developed will be illustrated with a case study: the *Lieue de Grève* territory, in Brittany, France. The objective of this case study is to help local stakeholders making decisions by determining which kinds of agricultural activities should be implemented in the studied territory, and where to locate them, to decrease their overall environmental impacts.

2. Spatialized LCA: a brief review

Environmental impacts caused by a pollutant emission depend on the quantity emitted, its properties, characteristics of the emitting source, and sometimes characteristics of receiving environment (Finnveden et al. 2009). In most LCA studies, the receiving environment is considered a standardized "unit world" with generic characteristics (Potting and Hauschild 2006). For some impact categories, such as climate change, this assumption holds true. However, some environmental impacts – such as acidification or eutrophication – vary depending on characteristics of the receiving environment and therefore on the location of the emissions. Spatialized LCA can take this variability into account in impact assessment.

In the literature, spatially explicit LCA has been addressed mainly at two levels: spatialized Life Cycle Inventory (LCI) and spatialized Life Cycle Impact Assessment (LCIA). While spatialized LCI is mainly based on the use of spatially differentiated databases and Geographic Information Systems (GIS) (Dufossé et al. 2013; Dresen and Jandewerth 2012; Engelbrecht et al. 2013; Geyer et al. 2010a,b; Núñez et al. 2010; Tabata et al. 2011; Tessum et al. 2012), spatialized LCIA focuses on spatial differentiation of characterization factors (CFs) for mid-point impacts such as acidification or eutrophication (Huijbregts et al. 2000; Krewitt et al. 2001; Potting et al. 1998; Shah and Ries 2009) or impacts of resource scarcity, such as that of water or soil (Boulay et al. 2011; Frechette-Marleau et al. 2008; Pfister et al. 2009; Saad et al. 2013; Verones et al. 2012). Spatialized LCIA has addressed mainly country, watershed or ecozone spatial scales (Pfister et al. 2009; Potting et al. 1998; Saad et al. 2011). Mutel et al. (2011) developed a computer program to combine spatialized LCI with spatialized LCIA. This technical solution still needs improvement, however, to be used by LCA practitioners. Spatialized LCA is increasingly common due to the use of GIS and the development of spatially differentiated CFs.

The debate about whether spatialized LCA reduces uncertainties in LCA studies remains open (Finnveden and Nilsson 2005). The amount of local data needed to spatialized LCA studies can indeed increase uncertainties in the LCI phase. Studies have shown, however, that using generic CFs can induce errors in LCIA, such as differences between site-specific CFs and site-generic CFs ranging from 2-4 orders of magnitude (Hellweg 2001; Hettelingh et al. 2005; Huijbregts et al. 2000). Therefore, taking the spatial variability of impacts into account seems necessary, especially when assessing the environmental burden of a territory.

3. Applying spatialized LCA to an agricultural territory

Few studies have used LCA to estimate a range of environmental impacts for an entire territory (e.g., Yi et al. 2007). Loiseau et al. (2013) developed a territorial LCA method but did not include spatial differentiation within the territory. We intend to develop a territorial LCA method that includes spatial differentiation within a territory for pollutant emissions and their associated impacts. For illustration purposes, we will apply this method to an agricultural territory.

To produce food or bioenergy for humans, agricultural territories induce environmental impacts that may vary depending on their surroundings where potential pollutants are emitted. Spatial analysis of agricultural territories has been performed outside the LCA domain for years. For example, models have been developed to predict spatially explicit dynamics of cropping systems (Leenhardt et al. 2010; Salmon-Monviola et al. 2012), spatial organization of land-use and rural territory dynamics (Benoit et al. 2012; Hinojosa and Hennermann 2012; Le Ber and Benoit 1998), or pesticide contamination risks in an agricultural watershed (Macary et al. 2014). The inclusion of a spatial differentiation in an LCA approach would provide a broader assessment of overall impacts.

3.1. Territorial LCA applied to an agricultural territory: the need for spatial differentiation

When estimating environmental impacts of a territory, one can consider it as a “black box” that interacts with other black-box territories via a variety of inputs and outputs. In this case, impacts of human activities are independent of location within the territory. We believe, however, that the territory should be considered as a system in which emissions occur at different places and impacts are influenced by the sensitivity of the receiving environment (Fig. 1). The need for spatial differentiation is especially relevant for land-use planning, where the location of certain activities may increase their environmental impacts. The method we develop thus aims to estimate spatialized impacts of an agricultural territory for land-use planning by coupling territorial LCA with spatialized LCA.

Although spatialized territorial LCA may require more data or time, it seems important to develop, especially for agricultural landscapes. It may help decrease impacts of a territory by estimating (1) what kinds of crop or livestock farming should be developed and where to locate them or (2) effects of land-use planning and how to minimize impacts of exchanges of resources or products with other territories, which may aid policy-making.

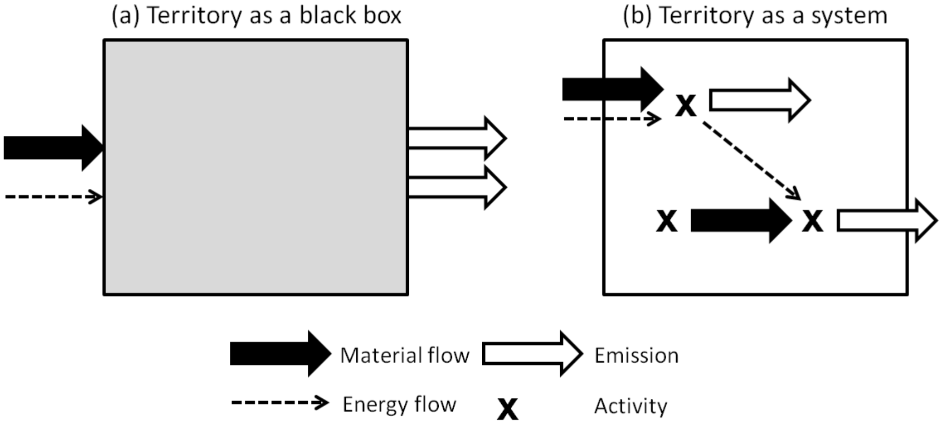

Figure 1. Two ways to estimate environmental impacts of a territory: (a) as a black box or (b) as a system whose activities, and inputs and outputs of those activities, are spatially explicit.

3.1.1. Defining the territory

The first step in a spatialized territorial LCA is to define the boundaries, functions, and issues of the territory under study. In agriculture, geographic boundaries of a territory are generally defined by the locations of farms and their fields but can be extended if entire agricultural sectors are studied. Next, defining territory functions and determining which of them is the most important is a crucial step in spatialized territorial LCA, since LCA is function-based. Loiseau et al. (2013) stated that territory functions can be mainly environmental, economic or societal. Functions of an agricultural territory may include producing food for human consumption or maintaining a certain living environment for the human population (Table 1).

Table 1. Examples of functions of agricultural territories.

Category	Functions
Environmental	Support biodiversity (e.g., reproduction and diversity of species)
	Support ecological processes (e.g., streamflow)
Economic	Produce revenue from agricultural activities
	Provide resources for industrial activities
	Support tourist activities
Societal	Produce food for human consumption
	Support livelihoods of the local population
	Maintain a certain living environment for the population (e.g., employment, quality of life)

The last step is to define issues at stake in the territory, such as to produce more food, to feed the local population or to decrease the amount of nitrate (NO_3) emitted to surface waters. Territory boundaries, functions, and issues, will influence which emissions and impacts will be studied (Fig. 2). They can be chosen iteratively (Goal and scope, Fig. 4) with the help of local stakeholders and experts, or with systemic analyses (pers. comm. Lynda Aissani, IRSTEA, Rennes, France).

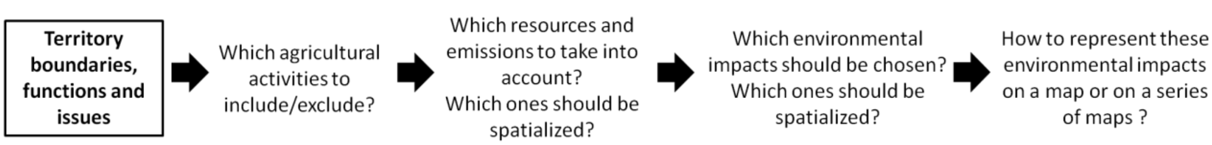

Figure 2. Effects of defining boundaries, functions, and issues of an agricultural territory on the LCA method.

3.1.2. Spatialized LCI

The next step is to spatialize the LCI, which requires distinguishing processes occurring within the territory from those occurring outside of it.

3.1.2.1. Processes occurring within the territory

Spatializing the LCI of the territory under study can be divided into three steps. The first step is to identify all agricultural and non-agricultural economic activities in the territory (e.g., with databases) and to geolocate them using GIS. It is important to include the non-agricultural activities, even though they will be examined in less detail. Since assessing the environmental burden of each human activity within a territory is too time-consuming, we propose developing a typology of agricultural activities, and use the typology proposed by Loiseau et al. (2013) for non-agricultural activities. The second step is to divide the territory under study into "zones of homogeneous environmental sensitivity" to each type of emission (e.g., nitrogen, phosphorous, and carbon compounds) using GIS. These zones (classified into an "environment" typology) should take into account environmental characteristics, such as the soil type, slope, and the distance to waterbodies. The third step is to calculate a spatialized LCI of activities within the territory. We propose to combine data from the activity and environment typologies to estimate spatialized emissions using appropriate models (Fig. 3). For example, nitrogen emissions from fields could be predicted using the TNT2 model (Beaujouan et al. 2002). Emission models should take into account pollutant fate as well as pollutant transport. Predicting transport is indeed crucial, since it will determine whether an emission (and thus its impact) will remain in the territory under study or travel to other territories. We propose performing LCA from the cradle to the territory gate, as suggested by Loiseau et al. (2013). A cradle-to-grave spatialized territorial LCA can be explored later.

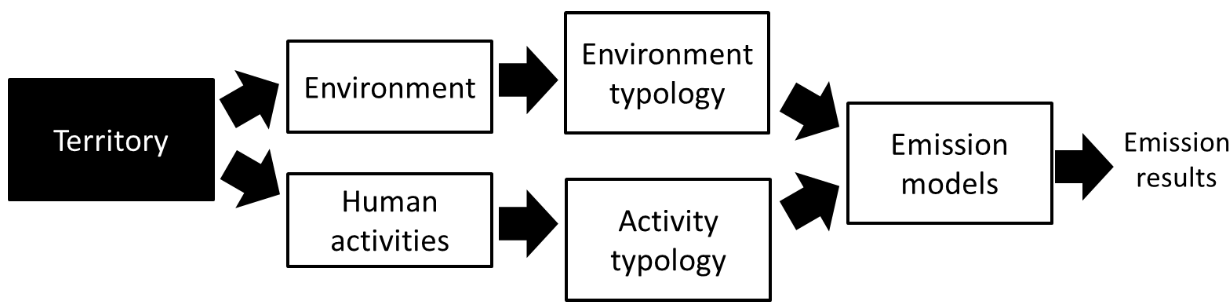

Figure 3. Spatialized life cycle inventory of activities within a territory

3.1.2.2. Processes occurring outside the territory

Indirect emissions from processes that occur outside the studied territory should also be included in the spatialized territorial study. The location of these emissions should be identified at least to the regional level, such as state or national level, when the origins of their contributing background processes are known

3.1.3. Spatialized impact assessment

Once emission locations, transport, and fate are determined, environmental impacts can be estimated. Spatialized environmental impacts will be estimated from spatialized emissions using spatially explicit characterization models. For each impact category, we propose assigning a CF to each zone of homogeneous environmental sensitivity. Environmental impacts in a given zone will be estimated from the emissions that end up there.

The appropriate spatial scale of each impact will depend on its impact category (Table 2). As a global impact, climate change does not need spatialization, unlike local impacts (e.g., eutrophication), which can be estimated more precisely with spatialization. For agriculture within a territory, we propose using a spatial scale at which a given impact is equivalent, such as a field for land-use impacts, homogeneous area of soil for terrestrial eutrophication and acidification, or homogeneous density of population for human toxicity (Nansai et al. 2005). A CF should be derived for each area of equivalent impact, which could be determined with GIS.

Table 2. Impacts categories and their spatial scales considered in spatialized impact assessment of an agricultural territory.

Impact category	Suggestion for spatial scale
Global warming	World
Eutrophication	
- Terrestrial	Homogeneous area of soil (e.g., type of soil, pH)
- Aquatic	Groundwater or surface water bodies
Acidification	
- Terrestrial	Homogeneous area of soil
- Aquatic	Groundwater or surface water bodies
Resource depletion	Location of resource extraction
Water consumption	Field
Land use	Field
Soil quality	Homogeneous area of soil
Energy consumption	Farm or homogeneous farm type
Human toxicity	Homogeneous density of population
Ecotoxicity	Ecozones

Larger spatial scales (e.g., watershed, region, or country) can be used for regions in which background processes occur. Importantly, the spatial scale of impact categories will depend on the impact category considered, the spatial scale of the environmental compartment (e.g., air, water, soil), and the type and quality of the environmental compartment (e.g., surface/ground water). If the agricultural territory under study is larger than the spatial scale of the impact categories, spatial differentiation within the territory seems necessary. In this case, site-dependent impact assessment can be used.

3.1.4. Interpretation

The final step, interpretation, can be a challenge, beginning with how to best map spatialized impacts to support decisions. We propose creating a detailed map of impacts within the territory along with a map of impact occurring throughout the world. To facilitate understanding, we recommend avoiding additional maps, showing only locations where impacts are the highest and reducing impacts on maps to a small number of scores. One objective of our method is thus to express spatially explicit environmental impacts in an optimal manner.

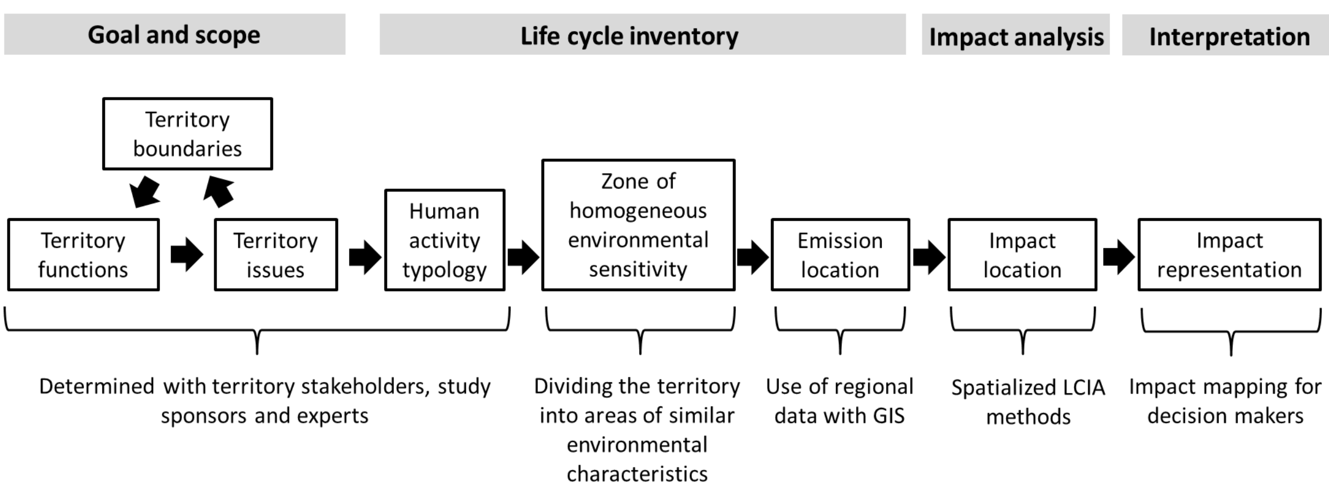

Figure 4. Spatialized territorial LCA steps

3.2. Case study: The *Lieue de Grève* watershed in France

Once methodological issues are addressed, we will apply the approach to the agricultural territory of the *Lieue de Grève* watershed in Brittany, France (Figs. 4 and 5). This territory was chosen because it is primarily agricultural with few urban areas, its main agricultural impacts define its boundaries (i.e., sub-watersheds of the

Saint-Michel-en-Grève Bay), and it has been studied before in research projects. Consequently, many descriptive data and a partnership with local stakeholders exist.

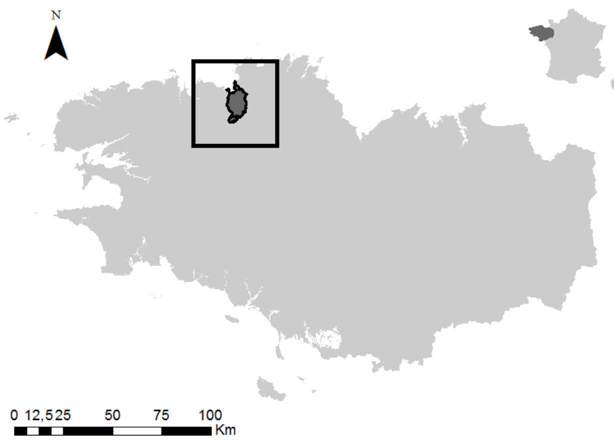

Figure 5. Location of the *Lieue de Grève* watershed (dark gray) in the region of Brittany, France.

Figure 6. Sub-watersheds in the *Lieue de Grève* watershed.

The *Lieue de Grève* watershed is composed of five sub-watersheds covering 120 km² and containing 12 towns and 13,500 inhabitants. Approximately 70% of the territory is covered by agricultural land, divided among 194 farms. Up to 85% of the agricultural land is used as dairy farms, while intensive pig/crop farming occurs on the eastern and western edges of the watershed.

The objective of the study is to analyze land-use planning scenarios to determine which kinds of agricultural activities should be implemented in the *Lieue de Grève*, and where to locate them, to decrease the territory's environmental impacts.

The main function of the *Lieue de Grève* agricultural territory is to produce agricultural products (mainly milk and beef), but this is common to all agricultural territories and does not represent a unique feature. One function with greater importance in the *Lieue de Grève* is the transfer of nutrients to the sea. Due to intensive agricultural practices and biophysical characteristics of land and bay, agricultural N and P that flow into the bay cause proliferation of green algae in the bay. Since this issue is important to the local population, we choose this function for the *Lieue de Grève* and define its geographic boundaries as its 5 sub-watersheds. In future research, the influence of changing the territorial function in LCA will be investigated.

The next step, spatialized LCI, is currently in progress, pinpointing with GIS locations of farms; their fields, emissions, and resource consumption; and environmental characteristics, using local datasets, on-farm surveys, and regional statistics. Transport and fate models will be used to determine the final locations of major emissions inside or outside the territory. The final steps will involve spatialized LCIA and interpretation of the spatialized results where scenarios of land-use planning will be compared.

4. Conclusion

Our method aims to combine spatialized and territorial LCAs to assess environmental impacts of an agricultural territory with a higher level of precision. This method refers to "spatialized territorial LCA". The key steps of spatialized territorial LCA are to: (1) define territory boundaries, functions, and main issues, since these characteristics will influence subsequent steps and results; (2) spatialize the LCI and LCIA using GIS; and (3) determine the best way to represent environmental impacts on a map to help stakeholders make decisions.

The objectives of this method are to: (1) give more accurate results than a territorial LCA without spatial differentiation, (2) help decrease impacts within a territory by determining which agricultural activities should be developed and where to locate them, and (3) help avoid or minimize impacts of exchanges of resources or products with other territories.

This method is still under development, and many methodological questions have yet to be resolved, particularly about the impact categories, and therefore the emissions, considered. The next step is to determine which indicators to use to define the zones of homogeneous environmental sensitivity. Another methodological issue is raised when combining these zones with the activity typology to determine the spatially differentiated LCI, particularly which emission models to choose to include spatial information in LCI calculations. It is also necessary to determine how to link the spatialized LCI to a spatially differentiated characterization method and how to best map impacts to aid decision making. For the latter, impacts occurring outside the territory must be represented.

The method will be applied to the *Lieue de Grève* territory, in Brittany, France, to assess its relevance from scientific and operational viewpoints. Ultimately, comparing spatialized and non-spatialized impacts will help us confirm – or disprove – the utility of the method.

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Carbon footprinting of dietary habits: the Meneghina Express project

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ABSTRACT

Meneghina Express is 44 day and 13,000 km long journey from Shanghai to Milan, covered via electric motorcycles as a representation of electric mobility and sustainable food production and consumption. The purpose of this journey was to promote sustainable mobility and also gather knowledge and best practices regarding food production and consumption along the route. In this context, the University of Bari developed a database that allowed the evaluation of the environmental burden related to various foods in terms of carbon footprint. Starting from the databases of the FAO, the carbon footprint of the consumed foods was estimated, drawing up an inventory of their life cycle. The result is a carbon footprint report, thanks to which it is possible to compare the "sustainability" of different foods encountered along the journey, with those of the Mediterranean diet.

Keywords: carbon footprint, diet, sustainable food consumption

1. Introduction

Various studies carried out during the last two decades have demonstrated that most of the human food chains are not sustainable because of their environmental burdens that occur during the various life cycle phases of the food systems. In fact the contribution of the food and drink sector to the total environmental impact deriving from private consumption, according to the European Science and Technology Observatory (ESTO) project on "Environmental Impact of Products" (EIPRO), ranges from 22 to 34% (Tukker et al. 2006). In the UK, food consumption is responsible for 19% of the overall 2008 national greenhouse gas (GHG) production. Similarly, according to the United Nations Food and Agriculture Organization (FAO), the livestock sector is responsible for 14.5% of the overall global GHG emissions (Gerber et al. 2013). In a search, over the last fifteen years, for some indication on more sustainable means for food production and consumption, various sustainability assessment instruments have been used, among which is the Life Cycle Assessment methodology (LCA). This has been predominantly used for the identification of the environmental impacts of food during its life cycle and also as a means of supporting environmental decision making (Notarnicola et al. 2012a).

In view of the above mentioned context the Meneghina Express project, object of the present paper, was developed. It entails a 44 day and 13,000 km long journey from Shanghai to Milan, covered via electric motorcycles as a representation of electric mobility and sustainable food production and consumption. This journey aimed not only at promoting sustainable mobility, showing that even adventurous journeys such as this one from Shanghai to Milan can be carried out by relying on renewable energy, but it also aimed at gathering knowledge and best practices regarding food production and consumption along the route. Since these themes are also among the main ones of the Expo2015 that will be held in Milan, the organizing committee of this exhibition sponsored the initiative.

For such a project, the University of Bari developed a database that enabled the evaluation of the environmental load related to the various types of food consumed during the journey. Currently, at global level for agro-industrial systems, there seems to be a growing interest in *carbon and water footprints* that represent "mono-indicators" that exclude the effects of other important impact categories such as eutrophication and eco/human toxicity. The current project also used the carbon footprint (CF) as means of evaluating the environmental sustainability of the food consumed during the journey; however a more comprehensive approach, that includes other indicators, is more desirable (Notarnicola et al. 2012b) and might be considered in the future. Specifically, starting from the UN Food and Agriculture Organization (FAO) databases, the CF of the single products was estimated thus creating a life cycle inventory of both primary (agricultural and livestock) and transformed products. This task was carried out for each nation crossed during the journey since the energy mix and other production variables imply different CFs for both primary and transformed products of different countries. The next sections of this paper discuss the methodologies and sources used for the construction of the inventory and the main results of the carbon footprint report of the journey.

2. The database for the food carbon footprint calculation

The starting point for the work regarding the calculation of the food CF database, was the analysis of the existing relevant data found in the scientific LCA literature regarding food products (agricultural and transformed ones). Also existing food databases (e.g Ecoinvent, FoodlcaDk and others) were preliminarily consulted together with data from environmental product declarations. Next, in a more specific manner, the FAO databases were used as data sources since they detail, for each country, the type of product and the relative land use, the use of fertilizers, the CO_2 emissions and other greenhouse gases (GHG) both for the agricultural phase and for livestock production (FAOSTAT 2013).

From the above mentioned data the CF for each single food product considered was calculated thus enabling the construction of the database of both primary and transformed products. The calculated CFs are intended as quantities of $CO_{2eq.}$ associated with the production of the specific food type and include other GHG such as CH_4 and N_2O. The phases included in the inventory analysis are the agricultural phase, the production and use of fertilizers and pesticides, the transport of auxiliary products and of the finished ones and the transformation. For livestock products the animal enteric fermentation and the type of manure management were also accounted for. All the results are referred to 1 kg of finished food product. For the conversion of all the GHG emission data to absolute values of $CO_{2eq.}$ the IPCC characterization factors were used (IPCC 2007).

The FAO database contains information on the following: total agriculture, combustion of agricultural residues, cultivated land, residues from cultivation, rice cultivation. All these inventories are calculated according to the *Tier1* approach indicated in the IPCC guidelines for the calculation of the national GHG inventories (IPCC 2006).

The inventories regarding the impacts of livestock, contained in the FAO database, are the following (also calculated according to the *Tier1* approach indicated in chapter 10 Vol.4 (IPCC 2006):

- <u>manure management</u>: the relative emissions are constituted mainly by methane and nitrous oxide deriving from the aerobic and anaerobic decomposition of the manure;
- <u>enteric fermentation</u>: this includes the emissions of methane produced mainly by ruminant digestive systems and in minor amount by those of non ruminant animals.

The CF of each product was calculated specifically for each country crossed during the journey. This is because the energy mix, the production factors and the efficiencies in terms of useful product vary between countries. The computation of the energy mix and its relative association to amounts of $CO_{2eq.}$ was based on the International Energy Agency databases (IEA 2013). As a result, for the non-transformed agricultural products, a database was generated, subdivided per each country, containing CFs of 136 products (not shown here for sake of brevity).

For the CF calculation for the various types of milk originating from different countries the yearly animal production yield illustrated in Table 1 was used.

Differences between intensive and extensive animal farming were also taken into account in the CF calculations. Specifically freely grazing animals are responsible for lower amounts of $CO_{2eq.}$ with respect to those of intensive systems which not only are accountable for emissions from enteric fermentation and manure decomposition but also for the production of fodder. However, intensive farming results less impacting in terms of CFs due to the higher production yields associated to the animals.

Table 2 illustrates the CF of milk form different countries. In some cases the results are particularly divergent. For example the CF of Mongolian milk is ten times higher than that of European milk which averages 1 $kgCO_{2eq.}$/kg. This is due to the extremely low yields in terms of product. Specifically, the FAO database reports a yearly milk yield per Mongolian bovine animal of 479.5 kg which is much lower than the Chinese (3,003.0 kg) or Russian yearly yield (3,857.2 kg). Such a result is well known and documented in literature as reported by Gerber et al. (2011). These results also are in line with the LCA report on GHG emissions from the dairy sector in which large variations in emissions between the different world regions have been estimated, with regional average emissions ranging from 1.3 to 7.5 kg $CO_{2eq.}$ per kg of fat and protein corrected milk (FPCM) [±26 percent]. Livestock systems in the temperate regions, mainly in industrialized countries, were found to have much lower emissions per kg of milk and meat than systems in the arid and humid zones in the developing countries (FAO 2010).

Table 1. Production yield in terms of milk produced (in kg) per animal over a period of one year for each country

	China	Mongolia	Kazakhstan	Russia	Moldavia	Ukraine	Romania	Serbia	Croatia
buffalo	543.3	0	0	0	0	0	0	0	0
camel	200.0	186.4	0	142.9	0	133.3	0	0	0
cow	3,003.0	479.5	1,893.0	3,857.2	3,405.2	4,175.0	3,775.9	2,921	4,246.1
goat	159.5	17.4	42.3	309.3	130.5	498.4	0	0	307.9
sheep	38.2	12.8	89.1	42.7	34.3	86.6	83.6	55.2	145.3

Table 2. The Carbon Footprint of milk for each country (in kg CO_{2eq}/ kg of milk)

	China	Mongolia	Kazakhstan	Russia	Moldavia	Ukraine	Romania	Serbia	Croatia
buffalo	2.23	-	-	-	-	-	-	-	-
camel	5.11	7.00	-	10.30	-	8.66	-	-	-
cow	0.65	9.60	1.22	0.68	0.71	0.51	0.67	0.76	1.02
goat	0.69	7.77	2.90	0.51	1.06	0.25	-	-	0.84
sheep	2.86	10.30	1.37	3.70	4.01	1.43	1.63	2.43	1.78

Global scale analysis has clearly shown that GHG other than CO_2, such as methane and nitrous oxide, are inversely correlated to animal productivity. In other words, compared to animals with low milk yields, animals that produce more milk, will consume more fodder and other foods, will undergo more enteric fermentation and produce more manure that will produce more GHGs. However when scaled to the amount of milk produced the CF is lower for animals with a high productivity. Hence increasing the productivity of livestock represents an effective strategy for the mitigation of the greenhouse effect associated to livestock breeding (Hristov et al. 2013).

Similar results are also obtained for meat production. In fact there appears to be a divergence in results in terms of CF also due to the differences in yields illustrated above.

Overall the database consists of 411 products for which a CF has been calculated. For 188 of these products the indicator was calculated for each country involved in the journey, whilst for the remaining 223 products the data was gathered from literature and existing databases.

3. The carbon footprint report

By using country specific food and energy mix data it was thus possible to evaluate the CF of each type of food eaten during the journey which varied considerably from country to country.

In China the nutrition of the team was based mainly on rice, pasta, eggs, vegetables and fruit with meat and small quantities of dairy products consumed on only two occasions.

In the Mongolian steppe, the nutrition was based mainly on dairy products and meat. In fact Mongols still practice nomadism and have large availabilities of such products. Specifically the nutrition included a type of pasta made from soft wheat, camel and goat milk, butter, sheep meat and a few onions and potatoes (the only vegetables eaten). This is mainly because Mongolia has very little arable land and thus produces few vegetables which are usually imported from China. This makes the Mongolian diet unbalanced, lacking vegetable foods and rich in proteins. This is also dependent on the fact that such a hyper caloric diet is necessary for surviving in a country with very rigid winters. In fact the capital city of Mongolia, Ullanbator, during the winter can reach temperature of -50°C. It thus appears that the climatic factor may be of considerable importance in the comparison between diets.

During the 11 days spent in Kazakhstan the team continued a diet similar to that undertaken in Mongolia, mainly based on meat with a lack of vegetable food. This depended on the route of the journey that involved crossing the northern part of the country through the city of Astana the second coldest capital of the world. In the southern part of the country the climate is milder and locally grown vegetables can be found.

During the 6 days spent crossing Ukraine the nutrition of the team was more continental, similar to that of northern Italy with a few variations based on fish from the Black sea.

A “carbon footprint report” was created in which, for each country, the average CF of the daily diet was calculated based on the food eaten. These values have been compared to those of the Italian Mediterranean diet. Presuming that the energy need during the journey could vary between 2000-2500 kcal per day, the CF of the Mediterranean diet varies from 4.9kg of CO_{2eq} to 6.1kg of CO_{2eq} per day.

Figure 1 reports, for each country, the averaged calculated CF of the daily nutrition. The horizontal line represents the Mediterranean average value.

As already mentioned the worst results are relative to Mongolia due to the high consumption of meat and dairy products. Specifically the team leader during the journey through Mongolia consumed 1.850 kg of vegetable food for a total of 0.54 kg CO_{2eq}., 0.55 kg of milk and other dairy products for a total of 8.64kg CO_{2eq}. and 0.3kg of meat for a total of 2.4 CO_{2eq}. as well as 0.2kg of transformed products for a total of 0.1kg CO_{2eq}.The highest contribution is from the dairy products and there is no major contribution from the electricity mix since most products (apart from butter) are consumed without any transformation. The meat consumption regards sheep and goat meat that typically have lower CF values than beef.

Summarizing, the results are higher than typical Mediterranean values due to the high consumption of milk and dairy products. The higher than average values for such products for Mongolia are due to the extensive nature of cattle breeding in such country. The best results in terms of CF regard China and are due to the high consumption of vegetables. In this particular case the nutrition of the team was biased towards such products due to safety food precautions that partially excluded milk based products from the diet.

The diets from the other countries, shown in figure 1, indicate a sustainability profile, in terms of CF, similar to the Mediterranean one and their largest contribution to the CFs depended on meat consumption.

These results confirm some findings of the extensive literature on diets. Indeed several studies compare different foods or compare various types of diets. A review by Heller et al. (2013) compared 32 LCA studies on diets. The results of all these studies pinpointed some key areas:

- in general, foods of animal origin show a worse environmental performance than those of plant origin;
- consequently vegetarian diets have a better environmental profile than other diets;
- there is a strong regional difference in food habits and production processes.

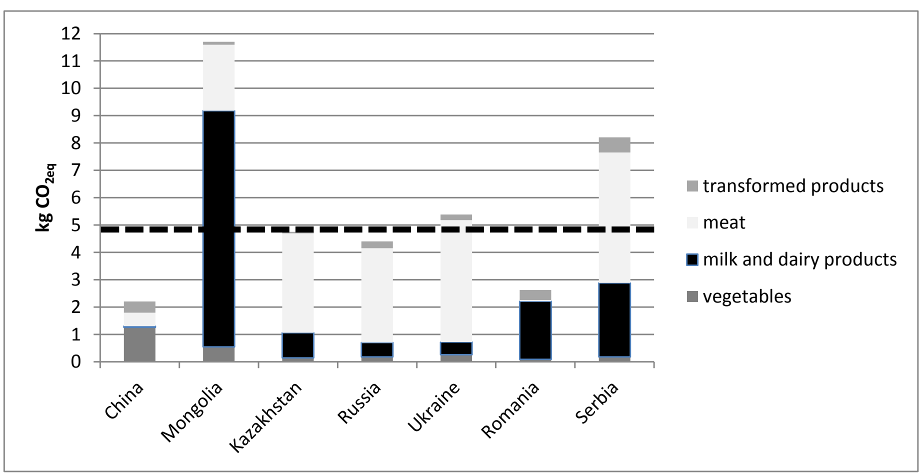

Figure 1. “Carbon footprint report”: CF of the daily diet calculated for the various countries

4. Conclusions

The Meneghina Express project, that obtained the Guinness world record for the longest distance travelled on electric motorbikes, represented a good occasion to focus on the environmental sustainability of food and diets. The desired results were reached thanks to the synergy between the exploration team and the University of Bari.

Regarding the creation of the database for the environmental sustainability assessment of food, it should be pointed out that it is a mere carbon footprint; the inclusion of other environmental aspects and impact categories was initially discussed but then excluded due to the lack of sufficiently reliable sources.

Overall the database is made up of the CF of 411 products. For 188 of these the indicator has been calculated for each country crossed during the journey, whilst for the remaining 223 products the data was gathered from literature and existing databases.

For many of the products the CF value varies considerably among countries due to the different production techniques and the their respective efficiencies.

Based on the data collected regarding the team's nutrition during the journey it was possible to generate a "carbon footprint report" of the various diets of the various countries and these have been compared to the Mediterranean diet. The results are of course representative only of this particular experience and do not represent national average values. Overall what emerges is that in the colder countries the diets are hyper caloric, based on meat and dairy product consumption and are less eco-compatible. In China where the nutrition was based mainly on vegetables the resulting CF is lowest hence counterbalancing the effects of the Chinese kWh which is based 79% on coal.

Concluding, food sustainability will continue to remain at the center of attention with the ever growing awareness of consumers, but it also needs technical instruments and reliable sources to find a scientific solution to the many problems connected to such a topic.

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GHG Emissions from an Aquaculture System of Freshwater Fish with Hydroponic Plants

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ABSTRACT

In this study, the GHG emissions of the farmed fish willow shiner (called Honmoroko in Japanese) were calculated including the GHG emissions for reforming the rice field to the aquaculture pond and expendable materials consumption on their production as well as the GHG emissions caused by energy on their production. Furthermore, the nitrous oxide (N_2O) volatilization from an aquaculture pond was estimated. As the results, the GHG emissions for the production of willow shiner were calculated as 3.62 t-CO_2e/t, which was correspond to 1,447 g-CO_2e/m^2/year, and the direct volatilization of N_2O was also estimated to be 572 g-CO_2e/ m2/ year. Nutritive salts removal experiment shows that SunPatiens flower has enough potential to remove inorganic nitrogen in the aquaculture pond. In other words, an aquaculture system with hydroponic plants has a capacity to make a clearance of N_2O from the aquaculture pond.

Keywords: GHG emission, aquaculture, cyprinoid fish, willow shiner, hydroponic plant, absorptive removal, nitrous oxide, volatilization

1. Introduction

There are many studies on the calculation of the GHG emissions for food, which are however calculated only for their production processes using the data of energy and expendable materials consumption. In this study, the GHG emission of the farmed fish willow shiner (called Honmoroko in Japanese) is calculated including the GHG emissions for reforming the rice field to the aquaculture pond and expendable materials consumption on their production. However, it is significant to know the direct volatilization potential of N_2O from the aquaculture pond caused by non-consumed feed and waste (ammonia and feces) of fish, moreover, the removal potential of inorganic nitrogen by hydroponic plant cultivated in aquaculture pond is also important to reduce the GHG emissions.

2. GHG emission of willow shiner

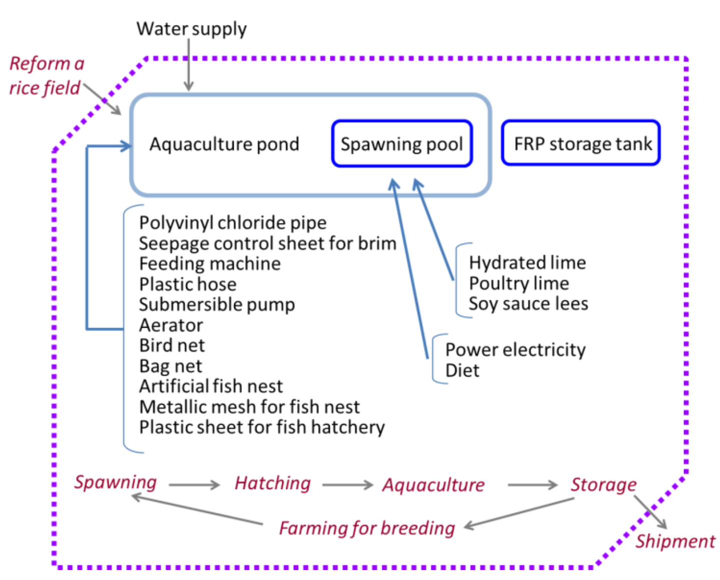

Figure 1. System boundary

In this study the amount of GHG emission for the farming of willow shiner is calculated including initial investment to reform the rice field to the aquaculture pond. The system boundary is illustrated in Figure 1. As water supply system differs according to farming region and area, it is excluded from the system.

The materials used for reforming rice field, aquaculture instruments, consumable goods, commercial fish feed, power consumption, hatching goods and storage materials are included in a system boundary. In order to obtain the amount of consumption of materials and energy, the balance of payment was used which is shown in a Tutorial Manual No.9 － Honmoroko Farming Technology－published by Saitama Prefectural Agriculture and Forestry Research Center Fisheries Research Insti-

Source: Organization for Regional Industrial Academic Cooperation, Tottori University

Figure 2. Examples of reformed rice fields in western Japan

tute. Additional interviews were performed in the Fisheries Research Institute in Saitama Prefecture and Honmoroko Farming Manual by Kusatsu city in Shiga Prefecture was also used as supplements for the GHG calculation. The rice field reform is performed shown in the figure 2 which are examples in Tottori Prefecture.

The 50-60 cm water depth is needed by digging a bit deeper the rice field and raise on the ridge. Waterproof sheets are laid on the ridge as shown in Figure 2. The amount of the GHG emission into units of CO_2 equivalent (CO_2e) is calculated according to the aquaculture schedule of the Table 1. 400 kg willow shiner per 1,000m^2 (10a) reformed rice field was produced in a year.

To calculate the GHG emission of the willow shiner farming, the official secondary data for Japanese Carbon Footprint Program are used firstly as the CO_2e emission factors and then the 3EID data (2000) derived from I/O analysis for Japan are used secondly. Table 2 shows the amount of consumption for durable /expendable materi-

Table 1. Aquaculture schedule

Late Feb.～Late Mar., Lime after dry the Aquaculture pond
Apr.17, Fertilization (Poultry manure ･Poultry manure)
Apr.26, Install a hatching pool
Oct.10 Install a fish tank

Jan.	Feb.	Mar.	Apr.	May	Jun	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.

Mid-Dec～Parent fish cultivation
Apr.28,29 Spawn
Nov.14 Harvest and storage
5/2 Install spawn in the hatching pool
5/8 Hatching
5/12 Set fry in the aquaculture pond

Table 2. The used primary unit and calculated GHG emissions (CO_2e)

GHG Intensity CO_{2e} emissions	Used amount (useful life)	CFP secondary data (t-CO_{2e} / t)	t-CO_{2e}	3EID monetary unit (t-CO_{2e} / a million yen)	t-CO_{2e}
Polyvinyl chloride pipe (drainage ditch, fish hatchery etc.)	35 kg (10years)	4.15	0.015		
Seepage control sheet for brim	55,740yen (10years)			4.564	0.025
Feeding machine 2	36kg (15years)	3.14	0.008		
Plastic hose	235 yen (10years)			4.564	0.000
Submersible pump 1	14,000 yen (10years)			3.901	0.006
Aerator 2	200,000 yen (10years)			3.901	0.078
Bird net	35,000 yen (10years)			10.314	0.036
Bag net	8,000 yen (10years)			10.314	0.008
FRP fish tank (6t)	220 kg (15years)	5.21	0.076		
Artificial fish nest	0.5 kg (5years)	6.31	0.001		
Metallic mesh for fish nest 2m^2	4,000 yen (10years)			2.876	0.001
Plastic sheet for fish hatchery	4kg (10years)	5.21	0.002		
Hydrated lime 200kg	200 kg (1year)	1.39	0.278		
Poultry manure 100kg	6,420 yen (1year)			2.906	0.019
Soy sauce lees 160kg	4,340yen (1year)			2.906	0.013
Commercial feed 670kg	670kg (1year)	0.321	0.209		
Burden charge	7,000 yen (1year)			3.612	0.025
Power electricity (270 days) kwh	1,320 kwh (1year)	0.479	0.632		
Supplies and all that (unclassifiable)	7,200 yen (1year)			2.144	0.015
CO_2 emissions (t-CO_{2e} /10a)		1.447			
CO_2 emissions (t-CO_{2e} / t)		3.62			

Table 3. The actual GHG emission of fishes considering wastage

	Wastage rate (%)	CO_2 emission (t-CO_2e / t)	CO_2 Emission from edible part (t-CO_2e/ t)
Common horse mackerel (sea culture)	55	3.672	8.16
Common horse mackerel (marine fisheries)	55	1.837	4.08
Saury (marine fisheries)	30	0.613	0.88
Ayu (Inland water fisheries)	55	10.555	23.44
Ayu (Inland water culture)	55	5.585	12.41
Rainbow trout (Inland water culture)	45	2.652	4.82
Willow shiner (Inland water culture)	0	3.62	3.62

als and energy to produce willow shiner per 1,000m^2 pond in a year and their CO_2e emission factors. Furthermore, Table 2 shows also the calculation results of GHG emissions (CO_2e) per 1,000m^2 pond per year. The amount of durable materials per year was obtained by dividing the whole weights by the years using them.

As the result, the CO_2e emission for the production of willow shiner is 1.447t-CO_2e/1,000m^2, which correspond to 3.62t-CO_2e /t-willow shiner. It is almost the same as those for other farmed fishes. However, as the edible part of willow shiner is 100%, while those of other fishes are about 50%, the actual GHG emission of willow shiner is thought to be smaller than the GHG emissions of other farmed fishes shown in table3.

3. Estimation of the GHG emission including Nitrous oxide (N_2O) from the aquaculture pond

670kg of fish feed, shown in Table2, was fed in the 1,000m^2 pond to produce 400kg of willow shiner per year. Non-consumed feed and waste (feces and ammonia) from fish are in the pond. The CO_2 emissions caused by them do not need to be counted, because the fish feed was produced by some plant and wild fish meal of phytoplankton origin, they are thought to be carbon-neutral. Moreover, the emission of CH_4 caused by non-consumed feed and waste from fish is negligible, because oxygen is always supplied by aeration in the pond for fish. However, we have to count the emission of N_2O caused by protein included in fish feed, because the global warming potential of N_2O is higher than CO_2.

As mentioned above, 670 kg fish feed was fed in the pond, which has 45.1kg of nitrogen because the 3 kind of commercial feeds shown in Table 4 were used. On the other hand, 400 kg of willow shiner has 11.2 kg nitrogen because the protein ratio of willow shiner is 17.5% according to the data of Standard Tables of Food Composition in Japan (2010) and nitrogen ratio of protein is 16% by nitrogen-protein conversion factor. It means 33.9kg of nitrogen was discharged into pond a year, which is the difference between 45.1kg contained in feed and 11.2kg of willow shiner.

Table 4. Ingredient labeling of commercial feed

Diet brand	Feeding period	The ratio of weight % to all diet (670kg)	Protein content (%)	Protein weight (kg)	Nitrogen weight (kg)
New carp mash	5/12～6/15	19.4	30	38.99	6.24
Carp juvenile C-2	6/1～7/31	34.3	45	103.41	16.55
Carp juvenile C-3	8/7～11/12	46.3	45	139.59	22.33
Total				281.99	45.12

Almost no research has been conducted to quantify N2O emission on aquatic production.

Ahn et al. quantified the annual N_2O emissions of 12 wastewater treatment plants across the United States and found that the N_2O emission can be as high as 1.80% of influent nitrogen. Using this ratio, the CO_2 equivalent emission (CO_2e) of N_2O for the production of willow shiner is calculated as follows. Global warming potential 298 is compliant with the (AR4).

$33{,}900 \text{ g} \times 0.018 = 610\text{g-N}$
$610 \text{ g} \times 44/14 = 1{,}917 \text{ g-}N_2O$
$1{,}917 \text{ g-}N_2O/ 1000\text{m}^2/ \text{year} = 1.92 \text{ g-}N_2O/ \text{m}^2/ \text{year}$
$1.92 \times 298 = 572 \text{ g-}CO_2\text{e}/ \text{m}^2/ \text{year}$

Adding this to the CO_2 emission, the total GHG emission (CO_2e) for the production of willow shiner is calculated.

$1{,}447+572 = 2{,}019 \text{ g-}CO_2\text{e}/ \text{m}^2/ \text{year}$

Around 2,000 g-CO_2e/ m^2/ year are totally emitted for the production of willow shiner. But it is very rough estimation because the conversion ratio of influent nitrogen to N_2O, i.e. 1.8%, was estimated by the studies of wastewater treatment plants which were probably different from aquaculture conditions.

4. An absorptive removal experiment of the inorganic nitrogen by hydroponic plants in a similar condition to an aquaculture pond

Hydroponic plants can absorb and remove the inorganic nitrogen in water. The potential to reduce the GHG emission from willow shiner aquaculture pond introducing hydroponic culture in the pond was estimated using the data of the following experiments by SunPatiens Salmon (*Impatiens* hybrid) flower, which showed the high absorption potential of nutrients in the water.

Hydroponic pots of SunPatiens and plant-free pot were prepared in small pools without aeration, which was called as “Pool condition”.

- Hydroponic pots of SunPatiens and plant-free pot were prepared with aeration condition, which was called as “Aeration condition”. It was the very similar condition to willow shiner aquaculture pond.
- 10% Hoagland’s solution was used as nutrient (NH_4-N: 10mg/L, NO_3-N: 10mg/L, etc.).
- Every week 10L 10% Hoagland’s nutrient solution was exchanged.
- Experimental period was for 7 weeks from 28th June to 11th August, but 5th week it was 6 days to be exchanged the 10% Hoagland’s solution and 7th week it was 3 days to be exchanged it (44 days), because SunPatiens salmon absorbed nutrient completely before 7th day in these weeks.

Table 5 shows Nitrogen absorption rate by SunPatiens Salmon. Aeration condition is similar to willow shiner pond. Mean absorption rate of total nitrogen (ammonium nitrogen and nitrate nitrogen) is 423mg/m^2/day. Figure 3 shows Nitrogen concentration changes of Hoagland’s nutrient solution, that an ammonium nitrogen concentration decreased more rapidly on the aeration condition plant-free pot than pool condition plant-free pot, and furthermore, the nitrate nitrogen concentration on the aeration condition plant-free pot was higher than pool condition plant-free pot. It is considered that aeration accelerate the nitrification and volatilization of ammonia. The 423mg/m^2/day might be over estimation because the ammonia volatilization is included. However, figure 3 also shows SunPatiens Salmon absorbed the nutrient of 10*l* Hoagland’s solution in a few days. As the period become progressively shorter in accordance with the biomass growth of the SunPatiens, and as expected the rate of absorption becomes larger, it would be used as the reasonable average value 423mg/m^2/day.

In the case of the mean absorption rate of total nitrogen i.e. 423mg/m^2/day, one plant has the absorption potential of 76g /m^2/180days. It means that 33,900g-N can be removed by 446 plants in 180 days.

Table5. Nitrogen absorption rate by the hydroponic plant (SunPatiens)

Pool condition

	Ammonium nitrogen			Nitrate nitrogen		
	Average loading (mg/m^2/day)	Mean absorption rate (mg/m^2/day)	Removal rate (%)	Average loading (mg/m^2/day)	Mean absorption rate (mg/m^2/day)	Removal rate (%)
SunPatiens (Salmon)	360.631	229.628	63.7	344.095	301.585	87.6
Plant-free	360.631	75.663	21.0	344.095	2.782	0.8

Aeration condition

	Ammonium nitrogen			Nitrate nitrogen		
	Average loading (mg/m^2/day)	Mean absorption rate (mg/m^2/day)	Removal rate (%)	Average loading (mg/m^2/day)	Mean absorption rate (mg/m^2/day)	Removal rate (%)
SunPatiens (Salmon)	360.631	125.538	34.8	344.095	297.373	86.4
Plant-free	360.631	192.691	53.4	344.095	9.652	2.8

Average loading, mean absorption rate and removal rare made an allowance for plant-free pot values

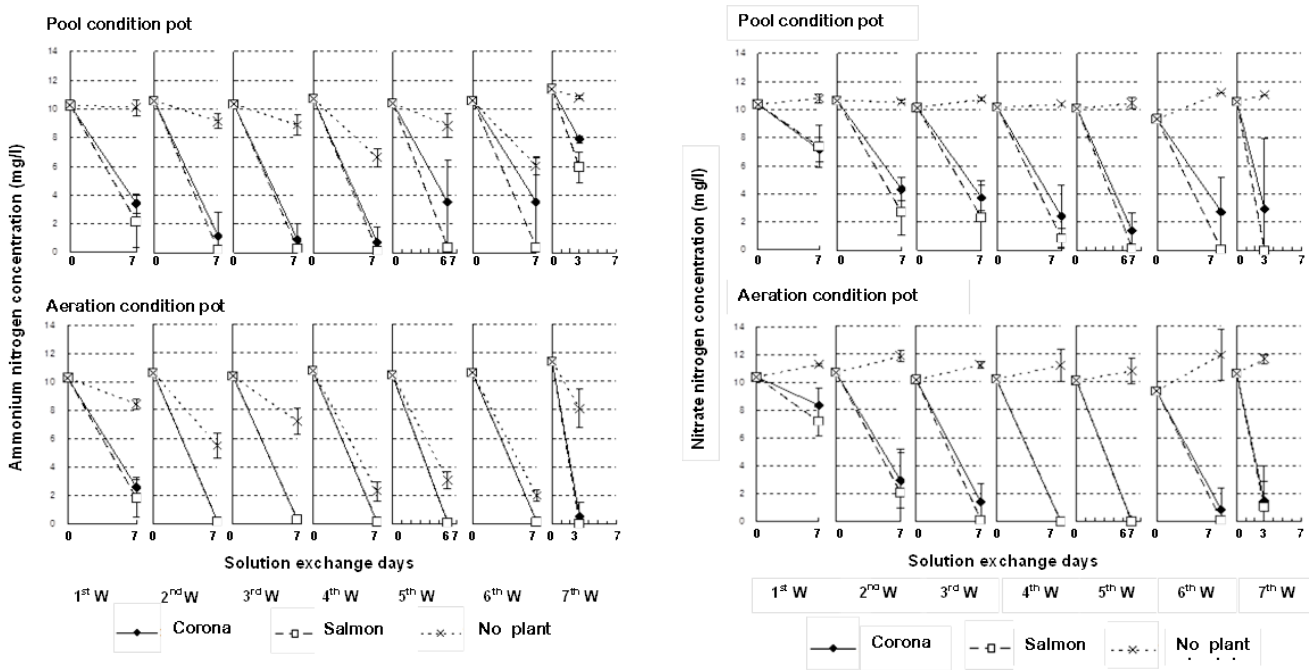

Figure 3 Nitrogen concentration changes of Hoagland's nutrient solution

5. Conclusion

A GHG emission for the production of farmed willow shiner in reformed rice field in Japan is around 2,000 g-CO_2e/ m^2/ year. It will be 3-4 times larger than the GHG emission of rice production of 728g CO_2e/m^2/year in Japan, which is calculated using 1.35kgCO_2e/kg-rice by the carbon footprint secondary data (B-J301001) and 539kg-rice /1000m^2 /year by the averaged rice annual crop (2013) of Japan.

The result shows the GHG emission caused by electricity consumption is the largest in willow shiner farming. The power is consumed for the aeration which is essential to freshwater aquaculture. The development of aeration technology which consumes little power is needed. The second largest emission source is fish feed. The fish feed production process of less GHG emissions should be developed, which might be the process using food waste.

In this paper, the absorptive removal potential of inorganic nitrogen by hydroponic flower, SunPatiens Salmon was estimated. The 33,900g-N, which was discharged in 1,000 m^2 pond for 400kg production of willow shiner in 180days, could be removed by 446 plants. As SunPatiens has the purifying effect of the aquaculture pond, the higher density fish farming can be achieved which could lead the less GHG aquaculture per the production volume.

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An evaluation of upstream assumptions in food-waste life cycle assessments

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ABSTRACT

The aim of this paper was to evaluate the tenability of the "zero burden assumption" for a waste stream with an economic value. Thirty comparative life cycle assessments (LCAs) addressing food-waste treatment were analysed and five questions were asked: (i) was the "zero burden assumption" used and what percentage of the waste system environmental impact was contributed by food-waste production? (ii) was there any indication that the waste had economic value? (iii) was it a comparative study? (iv) was the approach different between peer reviewed journals and commercial studies? and (v) if an environmental burden were assigned to the waste in each study, how might it be estimated? It was evident that the upstream 'zero burden assumption' is commonly followed in food waste LCA and no quantitative environmental impact is associated with the food waste resource, be it a comparative study or not. Few studies acknowledged that waste has an economic value. The argument for including the environmental impact of food waste and waste in general was reasoned on two fronts. Firstly, it was shown that the environmental impact of the waste itself may hold a large percentage of the overall system impact and should therefore be included. Secondly, with the valorisation of waste and its subsequent use as a feedstock in other systems that produce a product with an economic value, it is no longer directed to a biosphere sink, and should therefore not be called a waste as it stays in the technosphere as a resource. Future food-waste and waste LCA in general could follow the approach that was explored and quantify the environmental impact of waste by using a 'meta-waste-based accounting' approach.

Keywords: Life cycle assessment, zero burden, economic value, up-stream assumptions, technosphere, meta-waste-based accounting

1. Introduction

By definition, life cycle assessment (LCA) estimates the environmental impact of a product over its entire life from conception to disposal. The application of LCA for waste system analysis has been applied at macro, meso and micro levels in a large number of countries globally (Anderson et al. 2012; Khoo et al. 2010; Saer et al. 2013). A fundamental approach in LCA methodology is the modeling of a system being carried out in such a way that inputs and outputs to the system are followed from the 'cradle' to the 'grave'. This means that input flows should be drawn from the biosphere without human transformation, and outputs should be flows that are discarded to the biosphere without subsequent human transformations (ISO, 2006a). In the case of waste LCA a common approach applied in practice since the late 1990's is known as the "zero burden assumption" (Ekvall et al. 2007). It is applied when the waste coming into two comparative systems is regarded as the same across both systems and thus can be omitted from calculations (or assumed to have zero burden). In a scenario where there are differences in the amount of wastes coming into the systems under comparison the upstream boundary may have to be changed to include the impact of producing the waste (Finnveden, 1999). This is recognised as being difficult to do in practice, as waste is a non-homogeneous product (Ekvall et al. 2007). The aim of this paper was to evaluate the tenability of the "zero burden assumption" if the waste stream has an economic value, for example as a feedstock for nutrient recovery technology. Findings are discussed in relation to identified limitations, new developments and possible future research in the waste sector.

2. Methods

This study is limited to assessing upstream assumptions in food waste (FW) LCA. Other studies have reviewed the four stage procedure for carrying out FW LCA (Bernstad and la Cour Jansen, 2012). To evaluate the upstream assumption thirty LCA studies were considered (Table 1) from four continents, Asia (5 studies), North America (3 studies), Australasia (1 study) and Europe (21 studies). Criteria for accepting a study were established based on the following:

1. Include at least one nutrient recovery technology capable of accepting FW.
2. Adhere to LCA methodology.
3. Include global warming potential (GWP) as an impact category.
4. Have been published between the years 2000 and 2014.

A review matrix was then developed for the systematic review of the thirty papers, and the following five questions were then answered:
(ia) Was the “zero burden assumption” used and (ib) what percentage of the waste system’s environmental impact was contributed by food-waste production?
(ii) Was there any indication that the waste had economic value?
(iii) Was the study a comparison of more than one technology?
(iv) Was the zero waste assumption approached differently between peer reviewed journals and commercial studies?
(v) If an environmental burden were to be assigned to the waste in each study, how might it be estimated?

2.1. Reviewed studies

Table 1: Studies included in this review paper

Nr	Country	Technology	Waste type	Reference
1	Denmark	C	FW	Anderson et al. (2012)
2	Indonesia	AD/C/L	FW	Aye and Widjaya (2005)
3	Denmark	AD/C/I	FW	Baky and Eriksson (2003)
4	Turkey	C/I/L	MSW (FW)	Banar et al. (2009)
5	Belgium	AD/I/L	MSW (FW)	Belboom et al. (2013)
6	Sweden	AD/C/I	FW	Bernstad and La Cour Jansen (2011)
7	Italy	C/L	MSW (FW+GW)	Blengini (2008a)
8	Italy	AD/C/L	MSW (FW+GW)	Blengini (2008b)
9	Spain	L/C	MSW (FW)	Bovea and Powell (2006)
10	Sweden	AD/C/L	FW	Börjesson and Berglund (2007)
11	US	L/C	FW	Cabaraban et al.(2008)
12	Thailand	AD/I	MSW (FW+GW)	Chaya and Gheewala (2007)
13	Italy	AD/I/L	MSW (FW)	Cherubini et al. (2009)
14	US	AD	FW	DiStefano and Belenky (2009)
15	UK	AD/I/L	FW	Evangelisti et al. (2014)
16	Denmark	AD/I	FW	Fruergaard and Astrup (2011)
17	Italy	AD/I	FW	Grosso el al. (2012)
18	Spain	AD/I/C/L	FW	Güereca et al. (2006)
19	Singapore	AD/I/C	FW	Khoo et al. (2010)
20	South Korea	I/C/FP	FW	Kim and Kim (2010)
21	Denmark	AD/I	MSW (FW)	Kirkeby et al. (2006)
22	South Korea	I/L/C/FP	FW	Lee et al. (2007)
23	Australia	C/L	FW	Lundie and Peters (2005)
24	Spain	C	FW	Martínez-Blanco et al. (2009)
25	Spain	C	FW	Martínez-Blanco et al. (2010)
26	Italy	C/I	MSW (FW+GW)	Rigamonti et al. (2009)
27	US	C	FW	Saer et al. (2013)
28	UK	AD/C/L/I	MSW (FW)	Sonesson et al. (2000)
29	Italy	C	MSW (FW)	Tarantini et al (2009)
30	Ireland	C/L	MSW (FW)	White (2012)

C: Composting; AD: Anaerobic digestion; L: Landfill; I: Incineration; FP: Feed production; FW: Food waste; GW: Garden waste; MSW: Municipal solid waste.

3. Results

The results for question one, two and three are presented in Table 2.

Table 2: Results for question one to three.

Nr	1a	1b	2	3
1	No mention, study began with waste generated.	0%	Avoided fertiliser production.	Yes
2	No mention, study began with waste generated.	0%	Avoided electricity and fertiliser were considered. CBA carried out, comments that the end product would have an economic value.	Yes
3	No mention, study began with waste generated.	0%	Avoided electricity, heat and fertiliser were considered	Yes
4	No mention, study began with waste generated.	0%	Avoided fertiliser production was considered	Yes
5	No mention, study began with waste generated.	0%	States that an economic study using the concepts of life cycle cost would be conducted. Includes the generation of biogas.	Yes
6	No mention, study began with waste generated.	0%	Avoided electricity, heat and fertiliser were considered	Yes
7	Discusses 'zero burden' approach. Study began with waste generation	0%	Avoided fertiliser	Yes
8	Discusses 'zero burden' approach. Study began with waste generation	0%	Avoided fertiliser	Yes
9	No mention, study began with waste generated.	0%	Avoided fertiliser	Yes
10	No mention, study began with waste generated.	0%	Avoided electricity and fertiliser were considered	Yes
11	No mention, study began with waste generated.	0%	No avoided burdens were considered	Yes
12	No mention, study began with waste generated.	0%	Avoided electricity and fertiliser	Yes
13	No mention, study began with waste generated.	0%	Avoided electricity	Yes
14	No mention, study began with waste generated.	0%	Avoided electricity	No
15	No mention, study began with waste generated.	0%	Avoided fertiliser and electricity generation	Yes
16	No mention, study began with waste generated.	0%	Avoided fertiliser and electricity generation	Yes
17	Discusses 'zero burden' approach. Study began with waste generation	0%	Avoided electricity, heat and fertiliser were considered	Yes
18	No mention, study began with waste generated.	0%	No avoided burdens were considered	Yes
19	No mention, study began with waste generated.	0%	Avoided fertiliser and electricity generation	Yes
20	No mention, study began with waste generated.	0%	Avoided fertiliser and electricity generation	Yes
21	No mention, study began with waste generated.	0%	Avoided electricity	Yes
22	No mention, study began with waste generated.	0%	Avoided electricity, fertiliser and soybean	Yes
23	No mention, study began with waste generated.	0%	No avoided burdens were considered	Yes
24	No mention, study began with waste generated.	0%	No avoided burdens were considered	Yes
25	No mention, study began with waste generated.	0%	Avoided waste going to landfill	No
26	No mention, study began with waste generated.	0%	Avoided fertiliser production	Yes
27	Upstream processes discussed and excluded.	0%	Avoided peat production and NPK production	No
28	No mention, study began with waste generated.	0%	Avoided fertiliser production	Yes
29	No mention, study began with waste generated.	0%	No avoided burdens were considered	No
30	Discusses 'zero burden' approach. Study began with waste generation	0%	Avoided fertiliser production	Yes

CBA: Cost Benefit Analysis.

A limited number of commercial LCAs have been carried out for FW. From this limited analysis (question 4) it was seen that there was no difference in the approach taken for commercial FW LCA in comparison to that of the academic papers. The zero burden approach was followed for both types of analysis with four (13%) studies making reference to following the 'zero burden' approach, whilst the other twenty-six followed the approach but made no reference to it. Subsequently, none of the thirty studies (commercial or academic) associated a quantitative environmental burden to FW.

The FW entering each system in the thirty studies was different in each case, which means assessing its environmental impact in a cradle to grave system is impossible (question 5). One method for calculating the environmental impact of FW would be to follow the approach taken by Milà i Canals et al (2011) who assessed the footprint for the Knorr food portfolio. A bottom up approach was impractical in this case due the complexity of the product range, which is a similar challenge for FW LCA. The 'meta-product-based accounting' LCA approach that created sixteen product types, for example, "dry soup–instant" with an average recipe that does not exist in the market, but is a good-enough representation of the hundreds of variants of instant dry soups in the market could also be applicable to FW LCA where waste can be categorised into a number of sub-groups (vegetable, meat, grain, processed food and so on) and an environmental impact could be assigned. To illustrate this approach and the potential impact that FW could have in its simplest form, three food products (beef, tomatoes and pasta) were identified where verified LCAs have been carried out and only the impact GWP was considered (Table 3).

Table 3: GWP of three food types

Product	**kg $C0_2$ eq/kg**	**kg $C0_2$ eq/kg (Average)**	**% of waste**	**kg $C0_2$ eq/kg for % waste**	**Reference**
Beef	11-25.3	18.15	10	1.815	Casey and Holden (2005); William et al (2006)
Tomatoes	0.5-1.7	1.1	80	0.88	Hogberg (2010)
Pasta	0.85-2.5	1.675	10	0.1675	Barilla (2013)

If it was assumed that FW consisted of 80% tomatoes, 10% pasta and 10% beef a GWP of approximately 2.9 Kg CO_2-e per Kg of FW would be estimated (Table 3). This is a similar value to the estimated two tons of CO_2 per tonne of FW (European Commission, 2010).

Table 4: Adapted from Blengini, (2008a), GWP impact for FW composting system

Process	**Kg CO_2 eq/kg**	**Percentage**
Waste bags†	0.017	0.53
Road transport†	0.012	0.38
Waste processing†	0.045	1.41
Biogenic Emissions from composting†	0.156	4.88
Avoided Products and carbon sinks†	-0.10	3.13
Waste generation	2.8655	89.67
Total	3.1955	100

†Number are approximations taken from graphs in the original publication.

Taking Blengini's (2008a) research as a case study (FU is 1 kg of waste and has detailed breakdown of process GWP impacts) and including the embedded GWP impact of waste (Table 4) it can be seen that waste generation has a considerable impact at approximately eighty-nine per cent of the total.

4. Discussion

In relation to FW LCAs and according to the ISO standard (ISO, 2006) an LCA should be carried out in such a way that inputs and outputs to the system are followed from the 'cradle' to the 'grave'. It has been identified in this paper that this is not the case for LCA carried out for FW and waste in general. The 'zero burden' assumption has been followed for well over a decade in waste studies, comparative or not, and was seen in all thirty of the studies reviewed, with the result being that waste was not given a quantitative burden in waste systems when carrying out life cycle analysis.

All of the thirty studies included a nutrient recovery technology that produced a product that has an economic value that could offset some or all of the inherent negative economic value of the waste, however the majority of the studies made limited reference to waste having an 'economic value' as a resource, although twenty-five studies had an avoided burden associated to the replacement of a fossil material. The fact that this resource stream stays in the technosphere and does not enter the biosphere as a waste as per the definition of 'grave' by the ISO standard (ISO, 2006) means that it should be referred to as a resource (Figure 1).

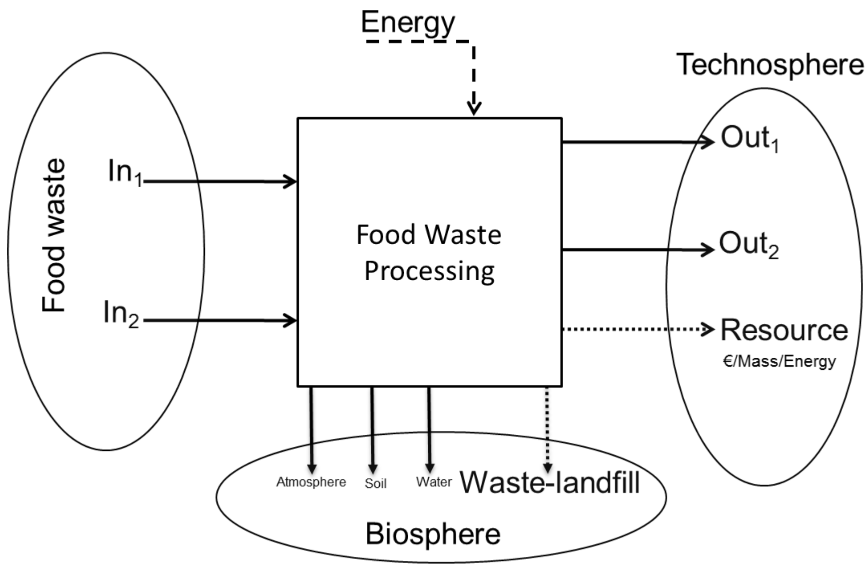

Figure 1: Waste verses resource

The term 'circular economy' is increasingly being used by governments and organisations (Mirabella, 2014) that are trying to find ways to close the loop on waste. This means it is used as a resource in other systems, such as a feedstock or raw-material replacement. In the case of biological nutrients found in FW this could be by composting or another nutrient recycling technology. However, the waste hierarchy defines that waste minimisation be the priority and in the case where reductions cannot happen it proposes that, reuse, recycling and recovery be achieved in that order. These two approaches can be seen as contradictory. In the case of adopting nutrient recovery technologies that would require investment and are privatised in many European countries, a return on investment and subsequent profit would be expected, which would come from the products produced (in addition to gate fee). Therefore it can be assumed that the owner of the facility would have no incentive to reduce the waste collected as this would decrease profit. This may not be the case in all locations across Europe, but with sustainability at the forefront of government policy and recycling rates on the increase waste can no longer be seen as a valueless resource, and must be dealt with in a manner that not only considers its economic value, but also the associated environmental burden.

This valorisation of waste, which has resulted in it becoming a feedstock for a number of technologies means that there is a strong relationship between its economic value and the amount that is available. In a case where 20% of total food produced is wasted (Figure 2) and becomes a feedstock in a recovery technology there will be a much stronger relationship between its economic value and the amount of the waste (20%) than that of the economic value and mass of the consumed food (80%). It can be argued that in this example where 80% of food produced is consumed that the economic value of this food or its disposal would have to increase significantly if this were to be the over-riding relationship between the amount of food wastage and amount food consumed.

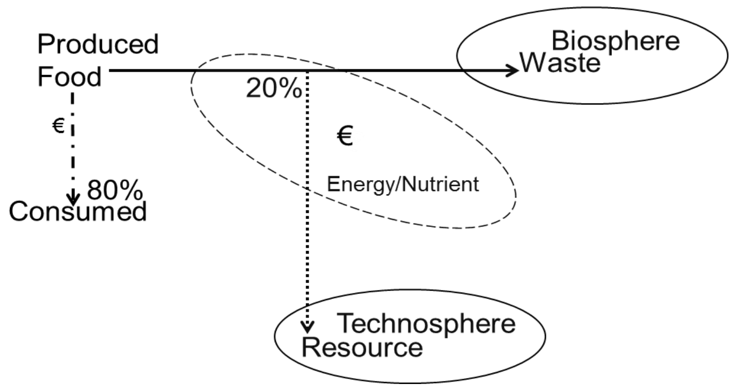

Figure 2: Relationship between the amount of waste and its value

5. Conclusion

In carrying out this review it was found that in response to the five questions asked: (1) the "zero burden assumption" was followed in all the studies thus no impact was assigned to food waste generation in all cases; (2) in five of the studies there was no reference to economic value, one study carried out a cost benefit analysis, whilst in the remaining twenty-four the product of the conversion process, i.e. compost or energy was identified as having a substitute value, but no value was mentioned for the food-waste itself ; (3) twenty-six of the studies compared two or more technologies; (4) the approach followed is the same for both academic journal and commercial studies; (5) the food waste entering each system in the thirty studies was different in each case, which means assessing its environmental impact in a cradle to grave system is very difficult.

It is evident that the upstream "zero burden assumption" is commonly followed in food waste LCA and no quantitative environmental impact is given to the food waste resource. The argument for including the environmental impact of FW and waste in general was presented on two fronts, firstly, it was shown that the environmental impact of the waste itself may hold a large percentage of the overall system impact and should therefore be included to show likely decreases if waste reduction occurs. Secondly, with the valorisation of waste and its subsequent use as a feedstock in other systems that produce a product with an economic value, it is no longer directed to a biosphere sink, and should therefore not be called a waste as it stays in the technosphere as a resource. Future FW and waste LCA in general could follow the approach that is suggested above and quantify its environmental impact by using a'meta-waste-based accounting' approach.

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An evaluation of introducing the OBEO caddy into the food waste disposal system in Ireland

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ABSTRACT

The objective of this study was to evaluate whether introducing the OBEO caddy to the home organic food waste disposal system would result in a lower environmental footprint than if the food waste was sent to landfill. A streamlined life cycle assessment (LCA) was carried out and the environmental impacts were quantified in terms of primary energy consumption (PEC), global warming potential (GWP), acidification (EP) and eutrophication (EP) potential. It was found that the OBEO caddy has the potential to greatly reduce the environmental and resource consumption impact of organic food waste and had a positive effect for PEC, GWP and EP. A negative effect for AP was found. Fill rate, and product design to achieve maximum fill rate need further attention and comparisons with plastic caddy's and recycled paper bags need to be undertaken to complete the evaluation of the OBEO caddy.

Keywords: LCA, Food Waste, Composting, disposal caddy

1. Introduction

Food waste is a global issue with an estimated one third of all food produced being wasted (UNRIC, 2014). In Ireland, a limited amount of food waste is captured in brown bins, which are sent for recovery in compost facilities. The Irish product design company OBEO has created an innovative, simple and fully compostable disposal caddy to facilitate food waste disposal in the home by making it clean and easy to undertake. OBEO's market research identified that by 2016 population agglomerations of more than 500 households would have access to a brown bin, which equates to a potential market of 1.2 million households in Ireland. The existing practice of landfilling food waste is no longer accepted as a suitable management option. The aim of this study was to estimate whether introducing the OBEO caddy to the home waste disposal system would result in a lower environmental impact than if the food waste was sent to landfill.

Figure 1. OBEO compostable Caddy

2. Methods

The streamlined environmental footprint and comparative analysis were performed in adherence to LCA methodology, standardised by ISO 14040 and 14044 (ISO, 2006a, b). LCA methodology has been used to evaluate and compare a large number of waste management technologies (Blengini, 2009; Kong et al. 2012; Martinez – Blanco et al. 2009). The LCA methodology consists of four phases: (1) goal and scope definition; (2) inventory analysis; (3) impact assessment; (4) interpretation, which were followed for this study.

2.1. System boundary and functional unit

The study is a comparative LCA, evaluating food waste disposal in landfill and in-vessel composting facility. This study did not consider the environmental impact of waste generation and its subsequent use as a resource. The upstream boundary was set at the waste collection stage. This is the approach followed in the majority of waste management studies, known as the "zero burden assumption" (Ekvall et al., 2007) and is applied when the waste coming into two comparative systems is regarded as being the same for both systems and thus can be omitted from calculations, or assumed to have zero burden (Finnveden, 1999). This method did not allow for the quantification of the food waste impact, but as this was a streamlined LCA it allowed the manufacturer to get a clear understanding of the potential impact their product may have.

The system under study consisted of the OBEO caddy manufacture and transportation to point of use; waste transportation from household to composting facility; treatment of waste in composting facility, the reference system was the collection of food waste and transport to landfill. The functional unit of the study was the disposal of 1000 kg of food waste.

2.2. In-vessel composting system

Once the food waste has been placed into the OBEO caddy it is consigned to the brown (organic waste) bin and sent for further treatment. The treatment process that was modelled in this study was in-vessel composting as it is the most suitable technology and is commonly used in Ireland. Full details of the composting process are outlined in Table 3. The data used in this study were taken from a number of LCAs carried out for Irish and European in-vessel composting processes. The composting plant modelled in had the capacity to process 15,000 t yr^{-1} of organic food waste.

2.3. Life cycle Inventory

The OBEO caddy consists of a paper bag which is housed in a cardboard (Kraftpack) case. Table 1 presents the inventory for the caddy manufacture and transportation.

Table 1: Characteristics of the food bag and kraftpak and distribution distances for the OBEO caddy.

Characteristics	**Food bag**	**Kraftpack**
Food bag mass (for one bag)	0.019 kg	-
Dimensions	200 x 115 x 390 mm = 0.00897 m^3	220 x 736 mm = 0.162 m^2
Volume/Mass	0.00897 m^3 = 8.97 litres Manufacturers assume 8 litres	0.162 m^2 x 0.283 kg/m^2 = 0.0458 kg
Manufacturer	Segezha Packaging	Kapstone paper
Location of manufacture	Southern Sweden	Illinois, USA
Delivery mode to Ireland	Ship & truck	Ship & truck
Distance	1637 km	5909 km

The OBEO caddy is held together with a glue and ink is used to place the logo on the Kraftpack. These items were not included in the analysis. Site specific data for the manufacture were not available, thus industry average data from the eco-invent and ELCD databases were used. It was assumed that these data were representative of the current, European produced paper and cardboard. Market research by OBEO defined typical food waste data (Table 2) used to design the OBEO caddy. Sensitivity analysis of the fill rate for the OBEO caddy was undertaken to evaluate the impact if not being completely full on environmental performance.

Table 2: Food waste data for the design of the OBEO caddy

Characteristic	Value
Food waste density range	343-515 kg/m^3
Food waste density average	429 kg/m^3
Mass of waste per OBEO	Maximum 2.3kg

It was assumed that the food waste for the reference system would be collected in a plastic bag at a rate of 10kg per bag. Table 3 presents the inventory data for the plastic bag.

Table 3: Characteristics of the plastic bag and distribution distances for the reference flow.

Characteristics	Plastic bag
Food bag mass (for one bag)	Assumed 10 kg
Volume/Mass	50 L
Location of manufacture	Assumed Illinois, USA
Delivery mode to Ireland	Ship & truck
Distance	5909 km

The compost facility data used for the study are presented in Table 4, whilst the data for Landfill is presented in Table 5.

Table 4: Compost inventory data used for the streamlined LCA.

Type of flow	Units	Quantity	Reference
Input			
Diesel	kg	4.73	Martínez-Blanco et al. (2009)
Electricity	KWh	50.5	Martínez-Blanco et al. (2009)
Food Waste	Tonne	1	Martínez-Blanco et al. (2009); Blengini (2008); White (2012).
Output			
Compost	Tonne	0.2	Martínez-Blanco et al. (2009); Blengini (2008); White (2012)
CO_2	kg	156	Blengini, 2008
Ammonia	kg	0.6	Blengini, 2008
Methane	kg	0.034	Martínez-Blanco et al. (2009)
VOC	kg	1.210	Martínez-Blanco et al. (2009)
Nitrous Oxide	kg	0.092	Martínez-Blanco et al. (2009)

The landfill was modeled using the process "landfill of biodegradable waste" from the ELCD database. Table 5 presents the typical flows for landfill.

Table 5: Landfill inventory data used for the streamlined LCA (ELCD, 2014).

Type of flow	Units	Quantity
Input		
Electricity	KWh	222
Food Waste	Tonne	1
Plastic Bag	g	300
Output		
CO_2	kg	62.6
Methane	kg	27.8
Nitrous Oxide	kg	0.00291

2.4. Life Cycle Impact Assessment

Four impacts (Table 6) were assessed, three followed the impact methodology CML 2001 (GWP, EP & AP).

Table 6: Impact Category, indicator and unit

Impact Category	Indicator	Unit
Global warming potential (Specifically - CO_2, N_2O, CH_4)	GWP_{100}	Kg CO_2 eq
Eutrophication potential	EP	Kg PO_4 eq
Acidification potential	AP	Kg SO_2 eq
Primary energy consumption	Energy Consumed	MJ

3. Results and Discussion

The results for the OBEO (Table 7) and landfill (Table 8) scenarios indicated that the OBEO caddy could lead to reduced PEC, GWP and EP, but increased AP (Figure 2).

Table 7: Impacts at composting per 1000 kg organic food waste if the OBEO caddy encouraged targeted disposal

	PEC (MJ)	GWP (kg CO_2e)	AP (kg SO_2e)	EP (kg PO_4e)
OBEO production	5.64	77	0.073	0.011
OBEO transport	0.53	3.5	0.0045	0.057
Waste transport	1.83	5.6	0.025	0.006
Composting process	700	150	35	0.22
Total	708	236	35.1	0.29

Table 8: Impact at landfill for 1000 kg of organic food waste

	PEC (MJ)	GWP (kg CO_2e)	AP (kg SO_2e)	EP (kg PO_4e)
Plastic Bag	4.6	5.0	0.013	0.001823
Waste Transport	52	4.0	0.019	0.0046
Landfill process	800	434	0.303	1.77
Total	856.6	443	0.335	1.776

Energy consumption might be reduced by 148 MJ for every tonne of waste composted rather than landfilled. There is approximately 200,000 tonnes of food waste generated in Ireland annually. If employing the OBEO caddy encouraged composting, this could result in a reduction of 29.6 million MJ of electricity consumption per year which is approximately 0.03% of Ireland's annual energy consumption. Reduction in GWP was calculated at 207 kg CO_2e ton^{-1} when using the OBEO caddy. This was mainly due to the amount of methane that is emitted during landfilling and the CO_2e associated with energy supply. Employing the OBEO caddy resulted in a potential increase in AP of 34.8 kg SO_2e ton^{-1} due to the composting process generating more documented acidifying emissions than the landfill process. It is worth noting that if this study was expanded to a full LCA then the result may be different because post-composting processes and more detail of landfill would be included in the full system model. A significant reduction in EP was observed with the introduction of the OBEO caddy and subsequent composting.

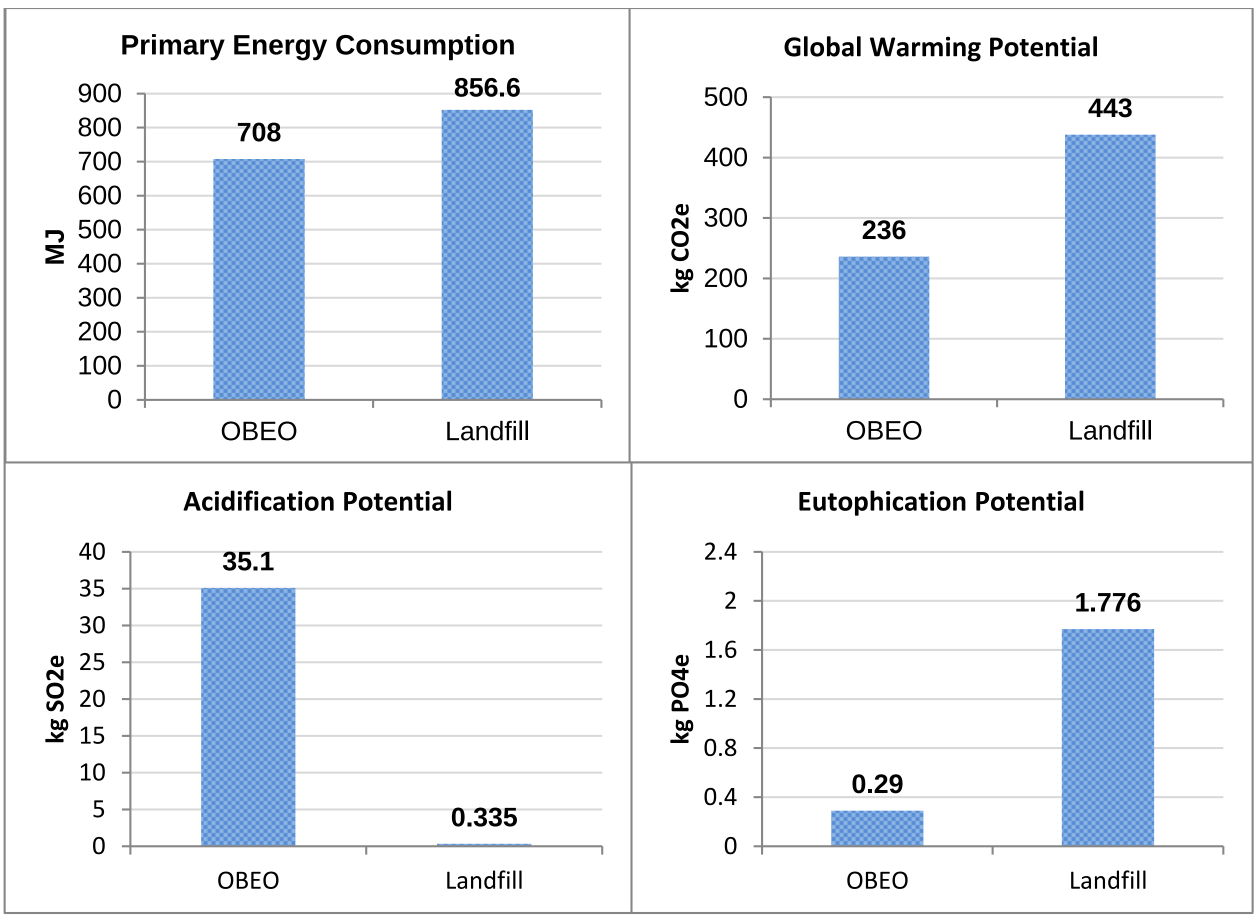

Figure 2. Comparison of the impacts for one tonne of waste composted with the OBEO caddy or landfilled

As this study was limited to a streamlined LCA, the main assumption tested was fill rate. It was assumed that the OBEO caddy would be filled to its maximum with a standard composition food waste (Table 2) when calculating the environmental impacts. The sensitivity analysis examined a 50% and 25% file rate. Reducing fill rate had a very small impact on PEC, AP and EP (data not shown), but there was a large impact on GWP (Figure 3). A 50% reduction infill rate caused a 74% increase in GWP. For 1 kg of waste to be captured approximately 0.4 OBEO caddies have to be produced and used at 100% fill rate. If the fill rate was 50% 0.8 OBEO caddies are needed and at 25%, 1.6 OBEO caddys. At a 25% fill rate the OBEO caddy, while encouraging composting would perhaps result in greater GWP because of the environmental impact of the manufacture and distribution stages in the product life cycle. Therefore to make sure the lowest GWP is realised, the design of the OBEO caddy must facilitate a high rate of food waste capturing, i.e. if an OBEO has a one day life span then its size should reflect the average amount of food waste disposed in one day. The sensitivity analysis suggested that consumers need to maximise use of the caddy to achieve optimum GWP benefit from composting using this system.

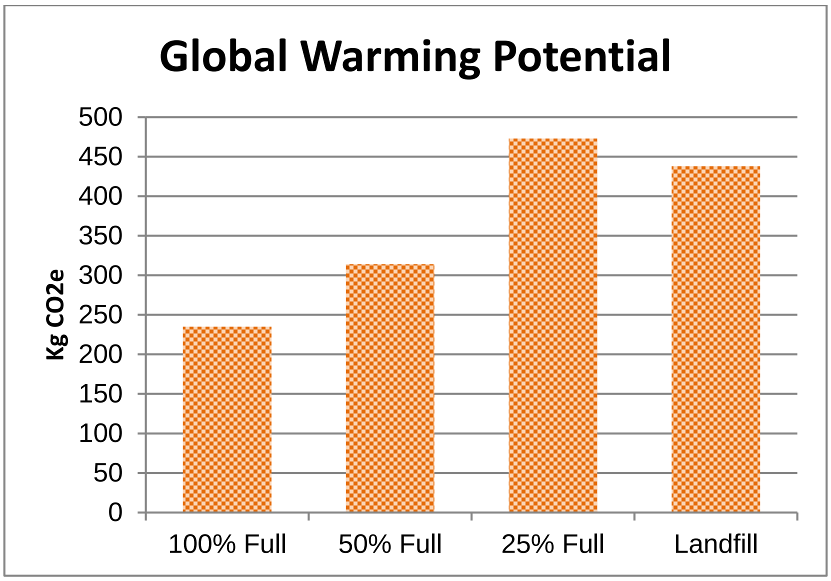

Figure 3. Sensitivity analysis of OBEO caddy to fill rate for GWP impact

As this was a streamlined LCA there was a limited comparison made between using the OBEO caddy to encourage composting and landfill. In reality there are at least two other approaches to food waste capture for composting that might be considered. Firstly the household simply captures food waste in recycled paper bags. As these are the most common bags for carrying shopping in Ireland due to the levy on plastic bags, it is a possible alternative. Secondly, the household captures food waste in a plastic caddy that is reused over and over. This latter alternative could also be lined with recycled plastic bags. It is clear from the design process of the OBEO caddy that simply using paper bags alone will not work well with food waste, which tends to have a high water content and thus paper shopping bags tend not to support wet food waste for transport to the recycling bin. While such an approach is likely to have less impact than the OBEO caddy if a zero burden assumption is used and a high fill rate, it is unlikely to encourage food waste recycling because of the mess and unreliability. This was the very reason for designing OBEO caddy in the first place. The use of plastic caddies is perhaps a serious competitor. In this case the life cycle burden will be diluted over many reuses of the caddy, but a cleaning impact (water use and EP may be important here) will arise. These alternative scenarios, and consumer behavior data on the relative encouragement that each scenario gives to separating food waste for composting need further research.

4. Conclusion

This study identified that the OBEO caddy has the potential to greatly reduce the environmental and resource consumption impact of organic food waste. The OBEO caddy has a positive effect for PEC, GWP and EP impacts but has a negative effect for AP impact, but this may be an artefact of the streamlined LCA. It was identified that fill rate, and product design to achieve maximum fill rate needs to be a focus of attention. To fully assess the value of OBEO from an environmental perspective a more complete LCA study should be undertaken.

It must be noted that the OBEO caddy by itself does not reduce the environmental impact, this only occurs when the food waste is diverted away from landfill and is composted. Thus, if the OBEO caddy was employed to collect food waste which was subsequently landfilled – there would be no environmental benefit, in fact there would be an increase due to its use.

An important outcome of this work is that if by using the OBEO caddy we can encourage more people to separate their food waste which would subsequently be composted in Ireland, then it should of course be used. For this study the important aspect of the lifecycle analysis is the waste treatment process, for Ireland composting or landfill. The OBEO caddy merely facilitates the transportation of the food waste.

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Using life cycle approach to evaluate trade-offs associated with payment for ecosystem services schemes

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ABSTRACT

PES (payment for ecosystem services) schemes generally operate locally, however it is important to consider their benefits from a life cycle perspective. This paper presents insight into the application of LCA to identify net environmental gains from CAMBI – a pilot scheme for sequestering soil carbon on-farm. A life cycle approach is applied by considering above ground GHG emissions from project implementation for a case CAMBI paddock in relation to soil C sequestration. The results suggested that increased soil C sequestration under CAMBI was sufficient to offset GHGs from project implementation and resulted in 0.58-0.7 t CO_2-e ha^{-1} yr^{-1} of avoided emissions under business-as-usual scenario. Further studies are needed to assess indirect impacts to identify environmental trade-offs from 1) land use changes in other areas of the farm or beyond; 2) farming input supply chain; or 3) other environmental impact categories. LCA can assist to identify net environmental gains from PES by addressing these issues.

Keywords: LCA, CAMBI, leakage, GHG emissions, Australia

1. Introduction

Agriculture is a dominant form of land management globally, covering nearly 40% of terrestrial land area (FAO stat 2010). Environmental impacts on ecosystems from agriculture are a function of: agricultural expansion into areas of natural ecosystems, agricultural intensification (higher yields from same or lesser land area), land use zoning schemes (which allocate land to restricted uses to ensure that natural ecosystems are not converted), and farming practices which can have both positive and negative impacts on the environment. Contrary to the belief that agricultural intensification and land use zoning reduces agriculture's impacts on ecosystems by halting agricultural expansion into natural ecosystems, studies suggest otherwise. Agricultural intensification or land use zoning in one geographical area potentially triggers compensating changes in trade flows of agricultural commodities and thus affect land use in other areas or countries (Lambin and Meyfroidt 2011; Pfaff and Walker 2010; Ramankutty et al. 2010). For example increases in import of cereals and wood products have been associated with countries that introduced conservation policies to keep land out of cultivation compared to countries without such policies (Gan and McCarl 2007; Mayer et al. 2005; Rudel et al. 2009). Thus in the era of globalization where there is worldwide interconnectedness of people and markets, and a greater separation between point of supply and demand, there is a greater need to understand and model outcomes of land use policies using a systems thinking approach.

Payment for ecosystem services (PES) is one such conservation tool that could benefit from a systems thinking approach to identify if such schemes are able to achieve net environmental benefits. PES offers monetary incentives to the landholders in exchange for some sort of ecosystem benefit in addition to food production. The idea is to bring otherwise free gifts of nature such as, clean air, climate regulation, pollination and so on, more in synchrony with the market, and in this way signal their importance in decision making. While PES schemes are increasingly being considered in sustainable agricultural practices both in developed and developing countries around the world, a need consistently reported is that of identifying environmental effectiveness of such schemes (Arrigada and Perrings 2009; Farley and Costanza 2010; Hajkowicz 2009; Tallis et al. 2008). A challenge which constrains the environmental assessment of local PES proposals is that of additionality and leakage. Additionality refers to the net impact on the biophysical provision of ecosystem benefits in comparison with the baseline[1] scenario or hypothetical situation where the PES scheme is not in place

[1]For example in Australia's Carbon Farming Initiative baseline refers to the carbon changes one would expect under business-as-usual condition that is those anticipated to occur in the absence of carbon policies or carbon projects. Setting the baselines is important as only carbon sequestered above and beyond the baseline will be counted for crediting purposes.

(Pascual et al., 2010). Leakage occurs when the actions to reduce environmental damages are merely displaced outside the geographic area of the project boundary typically resulting in offsetting or undermining environmental gains. For example, at the farm scale, leakage can potentially occur where project participants alter land use and farming practices in parts of their estate which are outside the technical boundary of the PES scheme. This could occur if land outside the PES scheme is managed more intensively to compensate forgone yields within the PES boundary.

In this paper a life cycle perspective - which is indicative of net environmental outcomes is applied in the assessment of CAMBI (Catchment Action Market Based Instrument) - a pilot project tested at the level of individual farmers and landholders for sequestering soil carbon (C) at the paddock scale. The aim of the CAMBI project is to design and test the cost effectiveness of a market based instrument (MBI) to support land holders adopt practices that increase soil C. This paper builds on previous and on-going work on modelling soil C levels through changed land management practices under CAMBI and considers the changes to above ground carbon emissions from project implementation. In this way the results indicate net potential gains from CAMBI intervention at the project boundary. This paper is organised as follows. In the next section a description of CAMBI and the methods to estimate net environmental impacts are presented. Section 3 and 4 reports and discusses the results of net environmental gains at CAMBI case paddock, followed by a final section on concluding remarks.

2. Methods

2.1. Description of CAMBI (Catchment Action Market Based Instrument)

Soil is a major sink of carbon; the size of the soil carbon pool and the annual flux of carbon passing through the soil are two reasons that soil organic carbon can play a significant role in mitigating GHG emissions. Soil C sequestration has been acknowledged to play an important role in reducing GHG emissions from agricultural systems in Australia (www.daff.gov.au/reducing-greenhouse-gas-fsheet.pdf). In addition to mitigating atmospheric GHG emissions, improving soil C levels has other benefits for increasing soil fertility, minimizing soil erosion, improving water holding capacity and ecosystem services and increasing farm productivity. With this background, the CAMBI pilot project, henceforth referred to as CAMBI, was designed and implemented in the Central West NSW to test the viability and cost effectiveness of a MBI to support landholders to adopt practices that increase soil carbon (Pearson et al. 2012). The premise is that potential changes to soil C occur if changes in land management are made. Accordingly, CAMBI has focused on assessing existing soil C levels under different combinations of land management, climate and soil landscapes. The predictions for soil C sequestration are based on the best available scientific knowledge and consider specific environmental conditions on specific soil types under different land holder defined actions and land uses (Murphy et al. 2012). Accordingly the CAMBI pilot scheme is also a test case to identify whether the current state of soil C science can support the practical implementation of MBI. The CAMBI pilot project was concentrated on the Cowra trough of NSW that has a temperate climate with a seasonally uniform rainfall distribution and porphyry-derived clay loam soils. The net soil C sequestered at the paddock scale from changed land management is the unit upon which payments were based. Contracts were finalized for eligible farmers through a reverse tender auction which provided best value in terms of soil C sequestered per dollar. The contract was between a farmer entity (proponent) and the Catchment Action NSW for a period of 5 years - from 2012 to 2017.

2.2. Data collection and analysis

Information was collected through a face-to-face interview from a CAMBI participant – a mixed farming enterprise with cropping and grazing of sheep on improved pasture. The farm is located in the Cowra region of NSW, Australia. Data was gathered on: 1) general information of the farm such as total land area, crop types, typical yields, irrigation availability, soil types and so on; 2) land management history of paddocks contracted under the CAMBI project – area, type of enterprise, typical calendar of operations, energy and material inputs used, typical yields; and 3) land management under CAMBI contract and the resources used therein.

Net environmental benefits of CAMBI intervention at the contracted paddock were estimated as the balance between GHG emissions from land management changes and predicted soil C sequestration rates expressed as t

CO_2-e ha^{-1} yr^{-1}. GHG emissions from cropping were modelled using LCA Simapro software (version 7.3) and both the Australasian LCI database and the Swiss Ecoinvent database (version 2.2) were used. Nitrous oxide emissions factor from N fertilization in cropping was estimated to be 1% (Australian Government 2011; IPCC 2006). Nitrous oxide emissions from crop residues (which are not burnt) is based on the National Inventory Report (Australian Government 2012) and takes into consideration factors such as yield, dry matter content and C:N ratio. GHG emissions from sheep grazing included GrassGro® modeled enteric methane and nitrous oxide emissions from excreted N and unutilized plant material. Historic climate information and data on-farm were used to model GHG emissions for an average year for the case farm.

Modelling soil C sequestration in CAMBI paddocks is part of the work based on SCaRP (Soil Carbon Research Program) – the largest and most extensive Australia wide program undertaken to measure stocks of soil carbon. As a part of this project a consistent method for sampling and a rapid and cost-effective way of analysing soil samples for soil organic carbon has been developed. Specifically, SCaRP-NSW was intended to assess the potential for agricultural land management practices (including emerging “carbon farming” practices such as the pilot CAMBI project considered in this study) to influence soil carbon on cropping and grazing land in NSW agricultural systems (Cowie et al. 2013). Sampling scheme for CAMBI project was developed to estimate the initial level of soil carbon in a paddock being assessed for a contract and its change after management actions have been undertaken (Murphy et al. 2012). This led to the development of a soil carbon metric that was used to assess rates of soil carbon sequestration for land managers which received payment under CAMBI contract.

3. Results

3.1. Historical GHG emissions on CAMBI paddocks (cropping)

The case farm is typically a mixed enterprise with cropping (wheat, canola, alfalfa) and sheep grazed on improved pasture. The property is not irrigated and is dependent on an average annual rainfall of 520 mm. Soil type is a combination of sandy and red clay loams. The paddock contracted under CAMBI is 80 ha, historically managed under a wheat-canola rotation for over a decade. Typically canola and wheat are sown in April and May and harvested in November and December respectively. The GHG emissions from wheat and canola in a typical cropping cycle are presented in Table 1. Typically wheat and canola yields are 3.5 t ha^{-1} and 1.75 t ha^{-1}. The stubbles for both wheat and canola are grazed and not incorporated into the soil prior to the next crop being sown.

3.2. Current GHG emissions under CAMBI contract

In summary, land management at CAMBI case paddock was converted from a wheat-canola cropping to sheep grazing on improved pasture. The improved pasture is sown once every 20-30 years with a mixture of grasses in combination with legumes. Typically, grasses included *phalaris, fescue, cocksfoot* and *prairie grass sp.* respectively, while the legume is *sub-clover sp.* Lime application of 1 t ha^{-1} was undertaken at the time of sowing, with a re-application frequency once every ten years. The stocking rate on the CAMBI paddocks was 3.8 ewes and 4.98 lambs per ha typically grazed for three months of the year. Total CO_2-e emissions for the above number of animals allocated over the time of year spent grazing on the CAMBI paddocks is estimated using GrassGro model to be 0.54 t ha^{-1} (Table 1).

Table 1. Net effect of CAMBI intervention at the paddock scale (project boundary)

GHG emissions before CAMBI - cropping			GHG emissions under CAMBI	Soil C sequestration
	GHG emissions			
Activity	**Wheat**	**Canola**	**Grazing**	**From cropping to grazing**
	t CO_2-e ha^{-1} yr^{-1}	t CO_2-e ha^{-1} yr^{-1}	t CO_2-e ha^{-1} yr^{-1}	t CO_2-e ha^{-1} yr^{-1}
Production and transport of fertilizers	0.25	0.4	0.10	
Production and use of diesel on farm	0.15	0.16	-	
Production and transport of pesticides	0.023	0.023	-	
Production of machinery	0.009	0.005	-	
Production of seed	0.008	0.002	-	
Soil nitrous oxide emissions from fertilizer application	0.140	0.116	-	
Nitrous oxide emissions from crop residues	0.007	0.003	-	
GHG emissions from livestock	-	-	0.44	
	0.58	0.70	0.54	1.27

3.3. Net environmental benefits at CAMBI paddocks

The predicted net changes to soil C sequestration under land management changes from cropping to improved pasture is estimated to be 1.27 t CO_2-e ha^{-1} yr^{-1} based on the soil C level monitored in the paddock and the equilibrium rate for the pasture system. The results suggest that soil C sequestration from changed land management under CAMBI paddocks did offset the emissions from project implementation (net sink of 0.73 t CO_2-e ha^{-1} yr^{-1}). In addition, changed land management from cropping to grazing under CAMBI further avoided 0.62 t CO_2-e ha^{-1} yr^{-1} (average emissions considering crop rotation of two wheat crops followed by a canola) which would have otherwise occurred under wheat-canola rotation. Thus, soil C sequestration from conversion of cropping to grazing was a net sink of GHG emissions considering the project implementation (grazing) and the avoided practices (cropping). This implied that the land management changes under CAMBI resulted in overall net benefits at the paddock scale.

4. Discussion

4.1. CAMBI - net environmental outcomes at paddock scale

The individual results for GHG emissions under business-as-usual scenario (wheat-canola cropping) are comparable with the published literature. For example previous work on carbon footprints undertaken in the same region of NSW reported between 0.56 to 0.70 t CO_2-e ha^{-1} (Brock et al. 2012) for wheat cropping as compared to 0.58 t CO_2-e ha^{-1} estimated in this research. Similarly for canola carbon footprints estimated in this study (0.70 kg CO_2-e ha^{-1}) are comparable with previous reported studies of 0.61 to 0.98 t CO_2-e ha^{-1} (Gan et al. 2012). Sheep grazing sector usually correspond to a higher above ground carbon footprints compared to white meats (pigs, poultry) and even with beef (Ledgard et al. 2011). The carbon footprint for sheep is dependent upon the management such as stocking rate, soil type, grass or grain-fed, climate characteristics and has been reported to vary widely ranging from 0.4 to 3 t CO_2-e ha^{-1} yr^{-1}(Edward-Jones et al. 2009). With the current stocking rate and the amount of time spent in the CAMBI paddocks used in this study suggested that carbon footprint for sheep grazing was lower than wheat-canola cropping on an area basis (Table 1). It is anticipated that similar number of stock would be grazed for an entire year in CAMBI paddocks in the future which will affect the GHG emissions; however the reason for short period of grazing (3 months) as was the case in this research was due to the establishment phase of CAMBI paddocks which were sown ten months earlier. On the other hand transitioning land management from cropping to grazing resulted in higher soil C sequestration than business-as-usual. This is consistent with the generally acknowledged understanding that soil C sequestration increased with management in the order: continuous cropping < crop-pasture rotation < pasture grazed by sheep and cattle (Chan et al. 2011). The result that grazing had net environmental gains at the CAMBI paddock scale is thus consistent with the above insights.

Based on the empirical work undertaken in this research, CAMBI intervention resulted in an overall net sink of GHG emissions at the paddock scale as compared to business-as-usual scenario. The system boundary for assessment used in this research was the paddock to coincide with the CAMBI project's system boundary. However, the paddock is not a stand-alone area and there is continuous exchange of information and resources between different areas of the farm. For example sheep grazed on the CAMBI paddocks for three months of the year; however the rest of the time they were dependent upon other areas of the farm. Hence it is important to understand the dynamics of GHG emissions outside CAMBI paddocks to identify whether CAMBI intervention has net environmental benefits at the farm scale. This means identifying 'leakage' in terms of GHG emissions from changed sheep numbers at the farm scale as compared to business-as-usual. Similarly information on land management changes to other areas of the farm from CAMBI intervention also needs to be identified.

4.2. Application of life cycle approach for assessing net environmental impacts of PES schemes

Although it is not unusual for empirical research to not develop hand in hand with the theory, policy design and implementation, the current state to measure environmental effectiveness of PES schemes is a cause of concern (Pattnayak 2010). For example the Australian Government's Carbon Farming Initiative in principle requires that the project participants discuss the issue of leakage. This implies that the project must not cause material increases in emissions outside the project's boundary, which nullify or replace the abatement that would otherwise result from the project. In theory this would involve estimation of a leakage deduction in the calculation of the net abatement or sequestration quantity (Australian Government 2010). In practice however, there is a need to include the criteria of LCA in the operational assessment of projects eligible under the CFI if the net benefits from such schemes are to be assessed. There are several potential advantages of life cycle approach implemented through LCA can offer in operationalizing environmental assessment of PES schemes. Firstly, LCA can handle environmental trade-offs across system boundary; this is especially relevant to agriculture where specific policy measures which are directed towards a single goal can lead to trade-offs elsewhere. For example the aim of CAMBI project is to test the implementation of MBI by paying farmers to adopt land management practices which enhance soil C sequestration. An application of life cycle thinking at the CAMBI paddock scale suggested that although for every ton of soil C sequestered there are 43% emissions from project implementation, there was a net benefit from CAMBI intervention. Secondly, LCA can assist to identify trade-offs with other environmental indicators such as water use and land use footprint; this enables overall environmental assessment of PES schemes. In this way, LCA can be a useful tool to identify the consequences of a decision such as PES intervention at the farm scale and beyond by considering off-site environmental impacts. Thirdly, life cycle approach can be potentially useful in discussing the issue of 'permanence' usually associated with sequestration projects. For example, the CAMBI participants have to comply with the project activities for a period of five years only. Potential land management changes beyond the temporal scope of the project using a life cycle approach can help to further identify net consequences of PES intervention in the longer term.

5. Conclusion

PES is an important and increasingly considered policy tool to encourage conservation from agriculture. However, continued poor evaluation in terms of additionality and leakage suggest paucity in the evidence of PES program's net environmental effectiveness. Life Cycle thinking applied through Life Cycle Assessment can offer advantages by considering net environmental gains from changed land management practices associated with PES schemes. The application of LCA suggests that GHG emissions from project implementation were not sufficient to undermine environmental benefits from CAMBI at the paddock scale. The CAMBI project resulted in being a net sink of GHG emissions as compared to business-as-usual scenario at the contracted paddock scale. However, the effect of CAMBI intervention on net environmental benefits at the farm scale and beyond needs to be considered in future studies under LCA framework. This would include the identification of tradeoffs with other ecosystem benefits from PES intervention such as reduction in food production or increased use of water and land resources. Thus future studies should reflect as closely as possible the consequences of changes to ecosystem services on other areas of the farm and elsewhere resulting from PES intervention. This implies using LCA to identify whether provision of ecosystem benefit at the farm scale is offset by a reduction in ecosystem benefit: 1) from project implementation; 2) in another area of the farm and beyond; 3) upstream in the farming input supply chain (from potential intensification); or 4) from trade-offs with other environmental impact categories.

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Environmental impact of processed tomato in France and in Turkey

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ABSTRACT

The FLONUDEP project carried out an environmental LCA « from cradle to grave » of processed tomatoes made in France and Turkey. We compared the environmental impact of tomato sauce made in France with French tomato paste, with that made with Turkish tomato paste. Data have been collected through surveys among a sample of farms (France=4; Turkey=4), processing plants (France= 2; Turkey=4), logistic organization (France=2), 1 supermarket and consumers (n=800), Findings show that French tomato sauce is slightly less impacting than the Turkish one for GHG emissions and human toxicity, whereas results are similar for eutrophication. Critical points are mainly packaging, energy used and steam production at plant level, fertilization and phytosanitary treatments at agricultural level, and finally, consumer behavior (shopping by car) and packaging recycling.

Keywords: Environmental impact, LCA, Tomato sauce, France, Turkey

1. Introduction

Studies on the environmental impact of agricultural products are now frequent; those on processed industrial products are rare. However, these are very useful as a European regulation regarding the display of the environmental impact of food products is currently under course. Industrials and supermarket chains (Casino, Auchan, and Leclerc) have already anticipated the regulation and present now a display of GHG emissions or water consumption related to the product's manufacturing. Environmental LCAs are fragmentary and often concern only one of the sectors of the food chain. However, several studies have been carried out on processed tomato products, with different system boundaries. Andersson's (Andersson et al., 1998) measured the environmental impacts associated with ketchup's life cycle, from "cradle to grave" and showed that the "hot spots" in the whole system were represented by packaging and food processing. Other two very recent studies (Del Borghi et al., 2014; Manfredi and Vignali, 2014) carried out an LCA on processed tomato from "cradle to the factory gate" (including packaging disposal) and from "cradle to the retailer" respectively , and highlighted the importance of packaging. Findings of the first study show that the impact of the agricultural stage is also considerable, whereas in the second, processing and transport to the retail center are also in cause.

The FLONUDEP project carried out an environmental LCA on the entire food chain, "from cradle to grave", for processed tomatoes in France and Turkey. The latter exports very few industrial products to France but has a huge potential. Turkey is in fact the 4th processed tomatoes producer in the world and holds 6% of the global market.

The purpose of the study is to conduct an environmental LCA of two processed tomato's value chains with the aim of identifying the environmental "hot spots" for each of them. The selected products are concentrated tomato paste and tomato sauce made in France and concentrated tomato paste made in Turkey and exported to France to be used in making sauce.

2. Methods

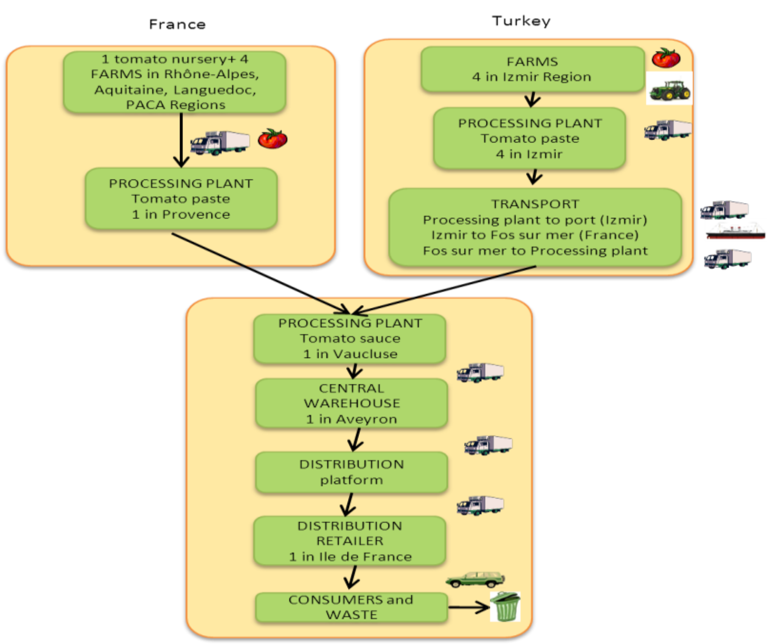
2.1. System boundaries

An LCA approach, from "cradle to grave" has been applied to both chains, that of French and that of Turkish tomato paste exported to France. For France, the system boundaries include: two tomato nurseries, four farms, a first (tomato paste) and a second (tomato sauce) processing plant, the warehouse, the distribution platform, the supermarket, the final waste, as well as transports between these. In Turkey, the study considered four farms, four first processing plants, as well as transport from the field to the plant and from the plant to France. Then the second processing plant in France, the warehouse, the distribution platform, the supermarket and final waste, as well as all transports between these. At agricultural level, allocation methods have not been necessary, as data was collected exclusively for tomatoes. Instead, at plant level, a mass allocation was applied in France and an economic one in Turkey; finally, at logistic and consumer level a mass allocation was applied. Logistics includes storage in the warehouse, the distribution platform, and the supermarket, as well as all the materials used for transport (further packaging), and type, consumption and fill rate of trucks used for transportation. The end of life of intermediate packaging (such as plastic bins and metal drums used to carry tomato paste to the second processing plant) and that of organic waste of fresh tomatoes, have not been taken into account, as the first are often recycled and waste from tomatoes are usually fed to animals. Finally, according to the consumer survey that was carried out during the study, consumers drive to supermarkets and recycle the tin can containing tomato sauce.

Comparison between the two systems taken into consideration in this study is possible, as much of the life cycle is in common (from the second processing plant onwards) (Figure 1). Furthermore, at plant level, whether in France or Turkey, the same flows necessary for tomato processing (energy, water and packaging-excluding its end of life-and storage) were considered in the inventory analysis. However, at agricultural level some differences exist: in France, the tomato nursery and the transport from the nursery to the farm have been studied, whereas this has not been possible for Turkey, as some producers grow the plants themselves.

Three impact categories were selected for the environmental measurements: GHG emissions, eutrophication and human toxicity. The first two are in fact among the criteria taken in consideration by the ADEME (French Environment and Energy Management Agency) regarding the regulation on the display of the environmental impact of consumer products, whereas human toxicity was chosen for its relevance. The environmental LCA was carried out using the SIMAPRO software and the CML 2 baseline V2.05 world method and Ecoinvent database.

Figure 1. System boundaries in France and Turkey



2.2. Functional unit

At agricultural level, in France and in Turkey, the Functional Unit (FU) is 1 Kg of tomatoes. At industrial level, the FU is 1 packed product at the plant's gate (1 metal drum or 1 plastic pouch for concentrated tomato paste, 1 can for sauce); for logistics and consumption the FU is 1 Kg of packed processed tomatoes. In spite of these FU differences between agricultural and post-agricultural levels, the system has been studied as a whole by converting all results in 1 Kg of packed processed tomatoes. The functional unit is therefore 1 kg of packed processed tomatoes using different reference flows for the different stages. We considered two cases: tomato sauce made with French tomato paste and the other with Turkish tomato paste exported to France.

2.3. Quality of data

Most of the data are issued from surveys conducted or from direct observations. To complete missing information, a literature revue and interviews with experts were conducted. In order to be representative, the 4 farms surveyed in Turkey have been chosen in the Egee region which produces 24% of Turkish tomatoes. We do not have the exact number of farms in this area but the choice is justified by the fact that the chosen farms supply the industrial plants under study, which buy 31% of Turkish industrial tomatoes. In France, industrial tomato production systems are very homogeneous, therefore the 4 farms chosen are considered as representative. The first processing plant buys 38.5% of the total industrial tomato production (SONITO, 2009) and the second processing plant is the first French firm of ready-made meals. The logistic taken in consideration is that of the two firms surveyed. Finally, at consumer level, the LCA is carried out with data from a survey among a sample of 800 people (men and women, 20 to 65 years old). It would have been desirable to carry out a sensitivity analysis; this had in fact been initially planned, but finally had to be postponed.

3. Results

The comparison between the two supply chains shows that environmental impacts associated with a can of 1 kg of tomato sauce as purchased by the consumer and then recycled is slightly lower for the strictly French chain, with regards to GHG emissions and human toxicity. The Turkish supply chain and the French one are identical for eutrophication (tables 1 and 2).

Table 1. Contribution analysis of the environmental impacts of 1 kg of processed tomatoes (sauce) produced in France from French tomato paste, by impact category

	Farm	Transport	Plant Tomato paste	Logistics	Plant Tomato sauce	Logistics to Supermarket	Supermar-ket	Consumers	TOTAL
GHG kg CO2 eq	0.0511	0.0063	0.0668	0.0032	2.2472	0.1219	0.0221	0.1800	**2.6987**
Human toxicity kg 1,4-DB eq	0.0307	0.0016	0.9727	0.0029	0.8478	0.0217	0.0358	0.4500	**2.3633**
Eutrophication kg P eq	0.0004	0.00001	0.0001	0.000001	0.0028	0.0001	0.0000	0.0002	**0.0037**

Table 2. Contribution analysis of the environmental impacts of 1kg of processed tomatoes (sauce) produced in France from Turkish tomato paste, by impact category

	Farm	Transport	Plant Tomato paste	Logistics	Plant Tomato sauce	Logistics to Supermarket	Supermar-ket	Consumers	TOTAL
GHG kg CO2 eq	0.0823	0.0259	0.3756	0.0086	2.2472	0.1219	0.0221	0.1800	**3.0635**
Human toxicity kg 1,4-DB eq	0.0247	0.0283	1.2374	0.0031	0.8478	0.0217	0.0358	0.4500	**2.6488**
Eutrophication kg P eq	0.0005	0.00003	0.0001	0.000011	0.0028	0.0001	0.0000	0.0002	**0.0037**

Regardless of the supply chain, the highly critical points are clearly the companies, particularly the 2nd stage processing plants. Indeed, it is at this stage that most of the emissions (¾) are generated, especially for GHG and eutrophication with the tin can and the energy used to produce steam (mostly natural gas) being the main causes. Regarding human toxicity risks, the 1st stage processing factory is also a critical point because of the metal drums used. Interestingly, the agricultural stage, whether in France or Turkey, has very limited responsibility with regards to all impact categories, with the exception of eutrophication (around 12% of the impacts).

At agricultural level, the impacts are quite similar between the two countries: GHG emissions are 1.6 times higher in Turkey, human toxicity is slightly higher in France, whereas eutrophication levels are practically the same. On the contrary, Turkey features much higher environmental impacts if we consider transport from the field to the first processing plant (an average of 70 km by tractor and 110 km by truck) for all 3 impact categories, as well as higher impacts for GHG emissions generated by the first processing plant alone. Regarding transport, findings highlight that rather than distance, it is the mean of transport that influences environmental impacts: for example, pollution (per kg of tomato sauce) due to transport between the field and the first processing plant (by tractor and truck) in Turkey is much higher (for example, 3 times for GHG emissions) than that between the Turkish processing plant and France (an average of 120 km by truck and 1225 km by ship). In France the shorter distance between the production sites and the processing plants (an average of 76 km) coupled with a rationalization of transportation of fresh tomatoes, makes GHG emissions 4 times lower than in Turkey.

Both in France and in Turkey, among technical operations, fertilization has the highest impact, compared with soil preparation, planting, protection, harvest and post-harvest work. This is true for 2 out of three impacts categories: global warming (F=62.7%, T=61%) and eutrophication (F=95%, T=93%). In France, an average of 106 kg of nitrogen, 134 kg of phosphorus and 223 kg of potassium are used per hectare, with great differences among the four farms. In Turkey, these are respectively, 133 kg, 90 kg and 184 kg. Yields are much higher in France than in Turkey, being 102.5 tons per hectare for the first and 72.5 tons per hectare for the latter. The second activity that impacts the most is soil preparation and harvest (36% of human toxicity and 30% of GHG in France). However, regarding human toxicity, phytosanitary treatments also play an important role in France.

In France, differences can be observed between the agricultural practices in the South-East and in the South-West. Due to different soil and climate conditions, use of fertilizers is higher in the South-East, whereas the opposite is true for pesticides. Therefore, eutrophication levels are 1.4 times higher for tomato production in the South-East. However, GHG emissions and human toxicity levels are similar.

The consumer stage also has a significant impact on the whole supply chain: 6% of GHG emissions, 19% of human toxicity and 5% of eutrophication. This is mainly due to the can recycling. Shopping also holds a part of the responsibility, as the majority of consumers drive to supermarkets. This generates a significant amount of impacts, especially with regards to global warming.

The main objective of an LCA is to compare the environmental impact of a product with that of another, in order to decide between the two. Therefore, we decided to compare the environmental impacts of two methods of tomato sauce preparation: the first one from fresh tomatoes and the second one from processed tomatoes. The consumer effectively choses between products that have the same function. At home it takes 2 Kg of fresh tomatoes, 30 minutes of cooking on an electric stove to obtain 1 Kg of sauce (according to our survey, the electric stove is the most common cooking method). The results show that at consumer level, both types of sauce have a similar impact on global warming. Differently, eutrophication is 1.5 times higher for home-made sauce while human toxicity is 10 times higher for sauce made with processed tomatoes. In the first case, the use of water is in cause, whereas for the second one it is rather the impact associated with the can.

4. Discussion

Very few studies take in account the whole system, including production, processing and distribution. The FLONUDEP project can be considered as the first of its kind in France. Results regarding Turkey are similar to others carried out in the same country (Karakaya and Özilgen, 2011).

Comparison with recent literature shows a certain consistency with results from the FLONUDEP project. In his study (Brodt et al., 2013), Brodt highlights the importance of transport rather than distance by making a comparison between a local supply chain of processed tomatoes and a national one. In this study, the choice of environmental impacts is also important because it can significantly change the results. Furthermore, an Italian study (Marletto and Silling, 2010) indicates that the impact of transport by car between the supermarket and home is a critical point which generates a significant amount of impacts. Another recent study in Turkey (Karakaya and Özilgen, 2011) highlights the responsibility of transport by tractor between fields and processing plants in generating impacts, as well as very different impacts depending on the energy source used in the plants. However, the supply chain stages which appear as the most impacting ones are not the same in this latter study as in the FLONUDEP study. This would require further research.

At farm level, if we refer to other LCAs for the same product (fresh field tomatoes) and with the same system boundaries, general results are similar: if eutrophication is higher in the production systems in Turkey (Karakaya and Özilgen, 2011), GHG emissions and acidification are higher in Spain (Martinez-Blanco et al., 2011). The French farms hold an intermediate position for all impact categories.

Table 3. Comparison of environmental impacts at the agricultural stage for industrial tomatoes (for 1kg of fresh tomato)

Category of impact	Unit	France	Turkey	Spain
GHG	kg CO2 eq	0.03424099	0.07951771	0.150
Acidification	kg SO2 eq	0.00058862	0.00049713	0.000888
Eutrophication	kg PO4 eq	0.00030962	0.00037501	0.000234

Source: Our study Flonudep, 2013; Karakaya and Özilgen, 2011; Martinez-Blanco et al., 2011

5. Conclusion

The advantage of such a study taking into consideration all the components of a system is to put into perspective the importance associated with some factors. Often agriculture is regarded as the most impacting sector. However results show that the main critical points are at plant level and more specifically packaging. It is therefore necessary to rethink packaging solutions and take into account their environmental impact associated with manufacturing and recycling. At plant level, two critical points clearly appear: the kind of energy used (there is a big difference between France which essentially uses nuclear energy and Turkey which uses fuel oil and gas) and the production of steam (with gas). A debate on production processes and “clean” energy is needed. . Furthermore, we should rationalize logistics according to environmental criteria rather than economic criteria. This would imply, for example, using local grouping platforms, closer to the production sites. We should also rethink transport for food products, as findings show that transport by truck in highly polluting. At agricultural level, as well as in all the other case studies taken in consideration, fertilization is the most polluting activity, followed by soil preparation and harvest for GHG. Phytosanitary treatment has also a significant role in terms of human toxicity. If ferti-irrigation is a partial solution, it would be useful to consider organic production or choose more energy efficient and less polluting agricultural equipment. Finally, it would be necessary to promote more “environmentally friendly” purchasing and consumption behavior among consumers. However, isn’t the modern supply and consumption system responsible? This is particularly true regarding consumption of ready-made, processed food, and supermarkets placed far out of the city centre.

The FLONUDEP research has been carried out on a limited sample of farms and processing plants in France and Turkey. Therefore, caution must be taken in interpreting results. However, the project highlights the critical points in the supply chain, and shows that according to environmental criteria, local products are not necessarily “better” than imported ones.

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Pass the salt please! From a review to a theoretical framework for integrating salinization impacts in food LCA

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ABSTRACT

Food LCA should include salinization. Salinization is a threat not only to arable land but also to freshwater resources. Nevertheless, salinization impacts have been rarely and partially included in LCA so far. First, a comprehensive overview of salinization mechanisms is presented and highlights its multiple causes, which affect soil and water, and ultimately human health, ecosystems and resources. Second, adopting the ILCD analysis grid, we analyzed the scientific relevance and accuracy of existing published methods addressing salinization in LCA. Although interesting, these seminal approaches are often incomplete with regards to both the salinization pathways they cover and their geographical validity. Third, we analyzed how to consistently integrate salinization within the methodological frameworks for impacts modeling in LCA, and raised questions to address towards a comprehensive integration of salinization.

Keywords: Salinization, Life Cycle Assessment, Agriculture, Soil, Water.

1. Introduction

Salinization is a global and major environmental concern. Although we commonly think this issue is limited to arid and semi-arid regions, it appears that no climatic zone is free from salinization (Rengasamy 2006). Furthermore, both agricultural and non-agricultural areas, both irrigated and non-irrigated lands can be prone to salinization (Wood et al. 2000). The FAO estimates that 34 million hectares of irrigated land are salt-affected worldwide, and an additional 60-80 million hectares are affected by waterlogging and related salinity (FAO 2011). Although reports of secondary salinization, i.e salinization due to human beings, continue to appear in the literature, there is a lack of recent assessment of the level of anthropogenic salinization (Flowers 1999). Salinization is a threat not only to arable land (EEA, 1997) but also to water resources (freshwater lakes and wetlands, rivers and streams), particularly in the arid and semi-arid regions of the world (Williams 1999). Salinity becomes a major issue in global agriculture when it adversely affects crop production (Rengasamy 2010).

As a global and multicriteria environmental assessment tool, Life Cycle Assessment should account for one of the major threats of food production. Research on including salinization in LCA is of high priority (JRC-IES 2011). Evaluating salinization impacts is particularly relevant in food LCA, because agricultural systems are both the main affected targets and causes of salinization. Sustainable irrigated agricultural production is seriously threatened by salinization (Aragüés et al. 2011).
So far, a few methods evaluated salinization impacts in the LCA framework: Amores et al. (2013); Feitz and Lundie (2002); Leske and Buckley (2003; 2004a; 2004b). However, these methods were rarely or never applied and focus on one salinization type or on a specific location. A comprehensive approach to salinization in the LCA framework is lacking.

The goals of this work are (1) to provide a comprehensive and structured overview of secondary salinization mechanisms and cause-effect chains, (2) to review LCIA methods modeling salinization impacts and (3) to identify methodological issues to build a consistent framework for salinization impacts in LCA.

2. Salinization: environmental mechanisms, cause and effect chains

2.1. A definition of salinization(s)

Salinization is the accumulation of salts, not exclusively NaCl as it is frequently assumed, but also many other types of salts such as carbonates, sulphates and other salts (calcium, magnesium and potassium) (Rengasamy 2010). Salinity is described as Total Dissolved Salts (TDS) (Leske and Buckley 2003). Although common ions are classified as non-toxic inorganic constituents, they may cause toxic effects at high concentration. Soil and water salinization are often studied separately: "*Salinization is the process that increases the salinity of inland waters*" (Williams 1999). "*Salinization is an accumulation in the soil of dissolved salts*" (Wood et al. 2000). But soil and water salinization are inter-related, water being the main vector of salts. Salts are conservative and resistant to degradation (Schnoor 2013) but they are mobile: they can either stay in a soil at a given location or they can migrate with water.

Soil salinization includes also sodization and alcalinization, while salt-affected soils include saline soils, saline–sodic soils and sodic soils (Ghassemi et al. 1995). Salinization is an accumulation of salts, and sodisation is an accumulation of sodium on the soil exchange complex causing soil clays dispersion, thus altering soil structure. Saline conditions are usually characterized by measuring the Electrical Conductivity (EC) and the Sodium Adsorption Ration (SAR) in soil extracts (Rengasamy 2010). EC measures the ability of water to conduct electrical current and is correlated with the total dissolved solids in soil water (Corwin and Lesch 2005) and thus salinity. SAR is the ratio of sodium ion on calcium and magnesium ions and is correlated with sodicity. Both salinity and sodicity are associated with more alkaline (basic) soils (Wood et al. 2000). Several salt-affected soil classifications exists (e.g. Rengasamy (2010)), and it is important to note that SAR and EC thresholds values are not the same in different soil classification systems (Rengasamy 2006).

Water quality in relation to salt abundance can be classified in 4 classes: fresh, brackish, saline and eventually brine water. On an operational point of view, farmers usually classify irrigation water according to EC and SAR measurements because it represents an easy way to estimate the salinization and alcalinization hazard of soil associated with its use (Richards 1954).

2.2. Causes of salinization in space and time

This paper does not consider natural salinization and focuses on anthropogenic salinization because LCA addresses impacts of human interventions. Salinization results from various causes which are often inter-related (Williams 2001). Salinization involves physico-chemical mechanisms at local scale and hydrological mechanisms at regional (catchment) scales. We distinguish for analysis between salinization associated with land use change and salinization associated with irrigation.

2.2.1. Salinization associated with land use change

Land Use Change (LUC) modifies hydrological processes and then water cycle at the catchment level. In particular, clearance of deep-rooted perennial native vegetation with high transpiration rates, and replacement with shallow-rooted crops with lower transpiration rates, will increase the water infiltration rate and mobilize salts stored in soil. Thus, the underlying groundwater tables can rise and reach the near soil surface in lowland, leading to temporary surface waterlogging and then deposition of salts through capillary action after evaporation (Williams 1999). In addition, the infiltrated salts increase the salinity of the aquifer. Many examples have been documented in Australia (Scanlon et al. 2007; Williams, 1999). Classical factors, as topography, acts on soil and water salinization at catchment scale. Specificities associated to LUC are directly linked to modifications of the water balance at the landscape scale: a modification of amounts of precipitations, a modification of groundwater table level, a modification of evapotranspiration rates, a modification of soil geochemical and hydrodynamic profiles, and salt stock variation in soil.

2.2.2. Salinization associated with irrigation

Salinization associated with deposition of ions - Irrigation is "*the salt concentration and mobilization machine*" (Smedema and Shiati 2002). The development of irrigation influences the local geohydrological regime, mobilizes salts stored in the underlying substrate, and contributes to the concentration of salts in land and water resources (Smedema and Shiati 2002). Irrigation water always contains some salts that may eventually accumulate in the soil, unless the irrigation management allows the salt leaching (Flowers 1999). However, the leached salts from the root zone may induce salinization of water bodies, such as the underlying groundwater, if drainage is not carefully managed (Mateo-sagasta and Burke 2010). Salts have a higher tendency to accumulate in semi-arid and arid areas because of the conjunction of low rainfall and high evapotranspiration rates (Marlet and Job 2006). The combined effect of the withdrawal of fresh irrigation water from a water body and the return of saline drainage water to this water body leads to salinity increase (Smedema and Shiati 2002). Furthermore, changing climatic conditions are worsening salinization with the use of low quality water to compensate the increased scarcity of freshwater (Duan and Fedler 2013). Irrigation's controls on salinization are those associated to local dependencies: salinization associated with deposition of ions through irrigation. Salts are also derived from fertilizers (Scanlon et al. 2007). Embedded key parameters are salt content in irrigation water, fertilizer use intensity, precipitation levels and irrigation doses, evapotranspiration rates, soil hydrodynamic profile, and salts reservoirs in the soil.

Salinization associated with shallow groundwater table or poor drainage - When an area is irrigated, the hydrologic balance is modified, and is often causing groundwater table rising. This rise may be noticeable after only a few years (Smedema 1993) cited by (Marlet and Job 2006). Areas having shallow water tables usually have soil salinity issues (Corwin and Lesch 2005). In this case, good drainage is required to avoid waterlogging and salts deposition through capillary action. More generally, poor irrigation management and inadequate drainage often lead to salinization and waterlogging (Wood et al. 2000). When the groundwater rising is saline, it may in turn induce salinization of some fresh waters (Williams 2001). Nested scales are involved in this salinization context: if spatial extend and structure of groundwater tables are regional, associated salinization processes are expected to act at a local scale; the key parameters are water table depth, drainage rates, salts content in irrigation water, precipitation levels, evapotranspiration rates, irrigation doses, and salts reservoirs in the soil.

Salinization associated with overuse of a water body: saline intrusion - In many coastal regions, excessive withdrawal of groundwater leads to marine intrusion: the decreasing aquifer table level and the proximity of seawater induces sea-water inflow in the aquifer. Some aquifers are already permanently salinized (FAO 2011; Flowers 1999; Scanlon et al. 2007). Deltas are also prone to marine intrusion when the freshwater flow of the river is reduced because of excessive water withdrawal upstream or the construction of impoundments (FAO 2011; Williams 2001). Sea-level rise induced by climate change is an aggravating factor of marine intrusion (FAO 2011). In non-coastal areas, saline intrusion may result from saline water transfer from a saline aquifer to an overused aquifer. Salinization associated with saline intrusion involves mechanisms at catchment scale; the key parameters are volume of freshwater withdrawal, distance to the coast, water body (river or aquifer) exploitation rate and presence of saline aquifer.

2.2.3. Salinization causes are often inter-related

Although we can establish a typology of salinization contexts, in many cases the situation is complex because salinization results from several causes. Many freshwater lakes, wetlands and rivers become saline because of the replacement of natural vegetation by agricultural crops upstream, together with the discharge of saline agricultural wastewater (Williams 2001). Groundwater salinization can be due to both sea-water intrusion and the agricultural return flows (Bouchaou et al. 2008). In addition, water and soil salinization are intimately related. The degradation of freshwater sources (surface or groundwater) has concomitant effects on the systems using these sources, and soil salinity affects in return water resources (D'Odorico et al. 2013).

2.3. Water and soil salinization damages to ecosystems, human health and resources

Salt-affected soils and waters have many effects and ultimately damage the areas of protection (AoP) as defined in LCA: ecosystems (or natural environment), human health and natural resources. Figure 1 depicts the salinization cause and effect chains, emphasizing the inter-relation between the multiple causes.

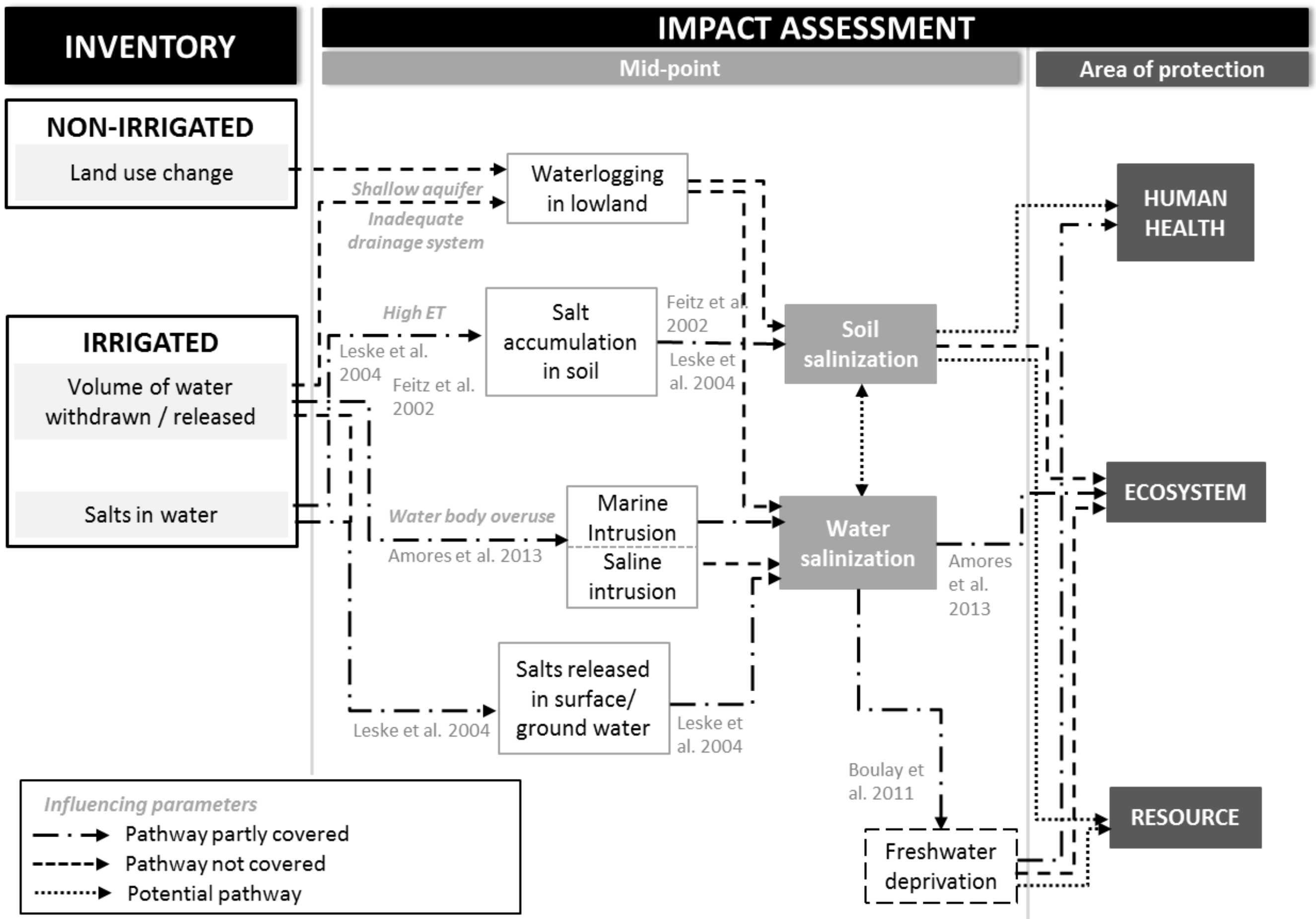

Figure 1. Human-driven salinization cause and effect chains and positioning of approaches proposed in the literature

Soil salinization not only reduces vegetation or crop growth, but also degrades land more or less permanently (D'Odorico et al. 2013). Salt-affected soils have a lower productivity through three potential effects on plants: i) reduction of plant water uptake by lowering the osmotic potential, ii) toxic effect by different ions depending on the pH, and iii) plants nutrients uptake imbalance (Flowers and Flowers 2005). Sodic soils also have indirect effects due to soil structure degradation, permeability reduction and possible waterlogging (D'Odorico et al. 2013; Rengasamy 2010; Suarez et al. 2006). Thus, by reducing crop yield, salinization of crop land could result in malnutrition for poor populations who rely on this production to feed themselves. Soil salinization is also considered as a driver of desertification because it affects terrestrial ecosystems and is closely related to land degradation processes such as soil erosion (D'Odorico et al. 2013). Land degradation might be considered as a damage to soil resource. It should be noticed that ecosystems may not lose diversity per se but simply replace the halosensitive biota with a halotolerant one (Williams 1999). Measures to reduce soil salinity and sodicity exist, but salinization can be irreversible: in arid regions where there is not enough freshwater available to leach the accumulated salts (Rozema and Flowers 2008), or in lowland areas of endorheic basins with shallow and saline groundwater (D'Odorico et al. 2013).

Salinization of a water-body not only affects the aquatic and riparian ecosystem, but also reduces the water availability for further use. An increase of water salinity causes change in species composition of algae, zooplankton, and benthic communities and leads to the disappearance of macrophytes and riparian trees (Schnoor 2013; Williams 1999). In addition, saline freshwater lakes, wetlands, rivers or aquifers are unfit to serve as supplies for domestic, agricultural and other uses (FAO 2011; Williams 1999), thus resulting in water deprivation for humans and ecosystems. This water resource quality alteration may be irreversible, for example for a permanently saline aquifer, and thus affects the water resource for future generations.

3. Review of salinization impact assessment methods in LCA

3.1. Overview of the methods

Three methods have been developed assessing salinization impacts in the LCA framework so far (Table 1). These approaches are either mid-point oriented (Feitz and Lundie 2002), end-point oriented (Amores et al. 2013), or near-endpoint oriented (Leske and Buckley 2003; 2004a; 2004b).

Mid-point oriented approaches - The soil salinization potential developed by Feitz and Lundie (2002) assesses the propensity of irrigation water to damage soil structure and the accumulation of sodium in the soil. The volume of irrigation water and sodium concentration are multiplied by a soil sodisation hazard characterization factor (CF). The soil sodisation hazard is assessed through the ratio between the EC threshold representing the limit of soil structure integrity for a given SAR, and the EC of the irrigation water. The SAR calculation is based on the irrigation water composition.

Table 1. Salinization impacts assessment methods overview

Publication	**Feitz and Lundie 2002**	**Leske and Buckley 2003;2004a; 2004b**	**Amores et al. 2013**
Inventory requirement	Irrigation water volume (L), [Na] (mg/L).	kg TDS released	Evapotranspiration of the crop: ET_{crop} (m^3)
Characterization factor (CF)	$CF=EC_{threshold}/EC_{iw}$ With: $EC_{threshold}=0.121xSAR+0.033$ linear equation representing the clay flocculation - dispersion threshold. SAR calculation requires: [Ca], [Mg], [SO4], [$CaCO_3$], pH, EC of irrigation water. If CF<1, no soil degradation hazard from sodisation.	Total salinity potential (TSP) for salt release in a given compartment: TSP= Σ potential effects on environmental target. With: potential effect $=\Sigma_i^N PEC_i-PEC_i^0$ / PNEC.M PEC_i=predicted concentration in the compartment during day i after an emission of total mass M; PEC_i^0=predicted concentration in the compartment during day i without an emission; PNEC=predicted no-effect concentration; N=number of days in the simulation	Change in PAF of species due to a change in groundwater consumption: - Fate Factor $=\Delta FGW/\Delta\ ET_{crop}$ x $\Delta C_N.V_N/\Delta FGW$ With: FGW=fresh groundwater inflow to Lagoon, C=salinity and V=volume of the lagoon, ET_{crop}=crop ET, Δ symbolizes the change between years. - Effect Factor $=\Delta PAF_{sal}/\Delta C_N.V_N$ $=0.5/HC50_{sal}$ With: $HC50_{sal}$=concentration at which ≥50% of the species are exposed to concentrations above their EC50
Indicator	$\sum_i CFi.[Na]i.Vi$ For irrigation water i. **Unit:** kg $Na+_{eq}$	TDS released x TSP **Unit:** kg TDS $_{eq}$	ET_{crop}. CF **Unit:** PAF.m^3.year, converted into species.year considering a 7.89 x 10^{-10} species.m^{-3} freshwater species density

Near-endpoint oriented approaches - Leske and Buckley (2003; 2004a; 2004b) developed a salinity impact category for South African LCA. They provide salinity potential CFs for salts release in atmosphere, surface water, natural surfaces, and agricultural surfaces compartments, and account for potential effects on aquatic ecotoxicity, materials, natural wildlife, livestock, aesthetic, natural vegetation and crop. Inspired by a risk assessment approach, salts fate factors are calculated with an atmospheric and hydrosalinity catchment model, thus assessing the predicted salt concentration in the different compartments. Effect factors are calculated using the predicted no-effect concentration for each environmental target.

End-point oriented approaches - Amores and colleagues (2013) evaluated the impacts on biodiversity associated with a salinity increase in a Spanish coastal wetland caused by the use of groundwater for agriculture. The water consumed by the crop (withdrawn from the aquifer) is multiplied by fate and effect factors. The fate factor represents the freshwater that lacks in the wetland system due to withdrawal, leading to increased concentration of salt due to increased sea water infiltration into the wetland. It is calculated from seasonal water and salts balances for the wetland Albufera de Adra. The effect factor is obtained from the fitted curve of the potentially affected fraction of native wetland species due to salinity. It is focused on plants, fishes, algae, and a crustacean.

Apart from the applicability test done by the authors themselves (Amores et al. 2013; Feitz and Lundie 2002), literature provides only one case study implementing Feitz and Lundies's method. Muñoz and colleagues (2010) calculated soil salinization potential as one indicator for soil quality impacts (besides soil organic carbon deficit), to compare the impacts of systems using treated wastewater, groundwater, or desalinated water for irrigation.

3.2. Critical analysis of the methods

We analyzed the methods against criteria of the ILCD Handbook procedure for methods analysis (JRC-IES 2011): completeness of scope, environmental relevance, scientific robustness and certainty, documentation, transparency and reproducibility, applicability and potential stakeholder acceptance.

Completeness of scope - Feitz and Lundie (2002) and Amores et al. (2013), focus on one single salinization pathway: soil sodisation and salinization from poor irrigation practices and seawater intrusion in a groundwater-fed wetland, respectively. Leske and Buckley (2003; 2004a; 2004b) cover several pathways of water and soil salinization induced by salts release. The methods are providing good methodological approaches to salinization impact modeling but with a limited geographical validity. Moreover, the methods cover only a part of salinization mechanisms within the pathway they are modeling. Figure 1 positions the contributions of these approaches on the global salinization cause and effects chains.

Feitz and Lundie (2002) do not account for waterlogging. Leske and Buckley (2003, 2004a, 2004b) do not consider salinization induced by a LUC or a saline intrusion, and Amores et al. (2013) do not consider groundwater salinization. All methods have site-specific CFs, emphasizing that salinization impacts are highly site-dependent; especially regarding the hydrology, the climate and irrigation water quality. However, their geographical extend validity is limited. The soil salinization impact of Feitz and Lundie (2002) depends on the validity domain of the electrolyte threshold curve which «*may not be appropriate for some soils*», and the estimation of the Sodium Adsorption Ratio of the soil drainage water is assumed for an Australian red-brown earth. Leske and Buckley (2003; 2004a; 2004b) fate factor is calculated with a South African catchment atmospheric deposition-hydrosalinity model, and the effect factor is based on the South African Water Quality Guidelines. Amores et al. (2013) fate factor is based on water and salts balance relying on the specific hydrologic functioning of the wetland and local hydro-climatic parameters, and the effect factor is based on specific native species of the Albufera de Adra wetland. It would be time-consuming and data-intensive for one who wants to apply the methods in other contexts.

Environmental relevance - Feitz and Lundie (2002) indicator is based on a relatively ancient approach but very common and generally well accepted. However, the soil type is not accounted for, although soil texture is a key parameter in the sodicity sensitivity. Indeed, sandy soils do not have soil structural problems caused by high SAR, whilst clayey soils are likely to be sodic with soil structural problems (Rengasamy 2010). Besides, the quality of the soil solution is buffered by slow physico-chemical mechanisms occurring over several years (Condom et al. 1999). Leske and Buckley (2003; 2004a; 2004b) fate model predicts environmental concentrations in all the compartments relevant to the calculation of salinity potentials. The land use distribution of the model is simplified through one single urban area, one single rural area and one single rural agricultural area. The calculated characterization factors for salts emissions onto the agricultural surface by far outweigh the CFs for releases into other compartments. This warrants further research to better model agricultural systems. The salts effect factors are based on the Predicted No-Effect Concentrations, a conservative approach compared with the HC50 (50% hazardous concentration), assuming that sensitivity of an ecosystem depends on the most sensitive species. Effect factors are not calculated as a function of the background salt concentration, except for aquatic ecotoxicity. Amores et al. (2013) fate and effect factor calculation are simplifying the salinization mechanisms. The effect factor is linear: calculated as the average gradient at the HC50 but does not account for background concentration. The Species Sensitivity Distribution are not based on EC50s describing the same effects (e.g., survival or growth inhibition). The fate factor is not utilizing any model but is based on simple water and salt balance equations.

Scientific robustness and certainty - These methods were published in peer-reviewed journals. Model uncertainty is not provided by Feitz and Lundie (2002), but is provided through a sensitivity analysis of the fate model by Leske and Buckley (2003, 2004 a&b), and through confidence intervals and CF standard error by Amores et al. (2013).

Documentation, transparency and reproducibility - Model documentation, characterization model and results published are available for the three methods. Reproducible for LCA studies located in areas where CFs are available.

Applicability - CFs are not straightforward to apply for the three methods. CFs have to be calculated by the practitioner because they are either irrigation-water dependent (Feitz and Lundie, 2002), or require re-developing the whole modeling approach because of its narrow geographical validity. The indicator units Na^{+}_{eq} and TDS cannot be compared with other methods as they are salinity-specific units. In contrast, the common end-point unit PAF from Amores et al. (2013) can be compared with other methods.

Potential stakeholder acceptance - The acceptance among LCA practitioners is limited with no or a single application (Leske and Buckley, 2003; Feitz and Lundies, 2002). Amores et al. (2013) method is young and remains to be implemented for validation in other contexts; further developments are required to ascertain if CF with global coverage can be calculated.

4. Perspectives toward a consistent framework for salinization impacts assessment in LCA

The purpose in this section is to analyze how salinization impacts should be modeled within the methodological framework of LCA. Answering this question raises several methodological issues: where to set the boundary between technosphere and ecosphere? What is the status of the AoP resource? A holistic approach is adopted to identify the methodological questions and risks of double counting to be addressed for future developments.

4.1. Methodological framework of LCA

We can distinguish two environmental impact modeling types in LCA: the first in relation to impacts due to the emission of a substance and the second in relation to the consumption of a resource. If tracing a substance in the environment has been historically addressed in LCA through the modeling of his fate and effect on the receiving environment, the modeling of impacts due to biotic resource consumption, especially for water and land resources, has only been studied recently (Finnveden et al. 2009).

Water and land as resources are inventory flows, which may be consumed or altered by the human activity under study. Modeling the damages caused by a resource use is complex because its function depends on the quality required by the user. Regarding water, if impacts from pollutant emissions into water are accounted for in LCA, impacts from water unavailability are not yet fully quantified (Boulay et al. 2011b). Boulay and colleagues (2011a; 2011b) developed a method to fill this gap: a functionality-based water impact assessment considering that water quality degradation can lead to water deprivation if not suitable anymore for specific users. In this method, the water quality is related to a functionality; assessing to which users the water is functional. This method assesses the impacts from water deprivation on the AoP human health. The same approach could be adopted to assess impacts of water deprivation on the AoP ecosystems, but the water categories should be based on ecosystems uses instead of human uses. Regarding the impact of water deprivation on the AoP resource, the framework proposed by the UNEP-SETAC “Water Use in LCA” (WULCA) working group considers that only fossil water use or renewable water overuse can affect the resources, thus water quality alteration is not addressed (Kounina et al. 2013). Yet, a permanently degraded freshwater represents a loss of water resource for future generation. There is no risk of double counting if we clearly define the AoP resource as the protection of a resource (in sufficient quality and quantity) for future generation, while the AoP human health and ecosystems reflect the protection of current people and ecosystems. The fact that the new IMPACT World^{+TM} LCIA methodology is considering an AoP "resource and ecosystems services" raises the question of the status of this AoP.

Although it has sometimes been considered as an impact indicator, land use is an inventory flow that leads to a group of impact categories, directly affecting ecosystems and resources, and indirectly affecting human health (Finnveden et al. 2009; Koellner and Geyer 2013a). The open research question is which midpoint categories should be defined to support an integrated impact assessment accounting for land use impacts alongside those from chemical emissions, water use and climate change (Koellner and Geyer 2013b). The UNEP-SETAC Life Cycle Initiative project LULCIA recommends for each impact pathway the determination of the optimal resolution of land cover types and regionalization, and the development of CFs for all relevant combinations of land use type and location (Koellner et al. 2013a). Several pathways were recently developed, within a structure of the LCIA in accordance with the typology of ecosystem services of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005). Indeed, the framework distinguishes between two intermediate end-points: biodiversity damage potential and ecosystem services damage potential. Brandão and Milà i Canals (2012) characterized land use impacts on biotic production potential (representing an ecosystem services damage potential) using deficit of Soil Organic Carbon (SOC) as an indicator. They developed CFs for eight land use types at the climate region level. Saad et al. (2013) address land use impact on three ecosystem services: erosion regulation potential, freshwater regulation potential, and water purification potential. They calculated CFs per land use type (seven) specific to each biogeographic region (tested at different regionalization scales), using LANCA® model which assesses the influence of different land use activities on soil ecological functions. Developing such CFs requires the availability of land-use-specific and biogeographically differentiated data on the indicator selected: SOC for Brandão and Milà i Canals (2013) or soil properties, landscape and climatic conditions for Saad et al. (2013).

4.2. Perspectives for integrating salinization in LCA

Regionalization - The fate of salts depends on climate, agricultural practices, soil, and hydrological context, and their effects depend on the sensitivity of the target (species, capacity to desalinate water). Therefore a regionalized approach is paramount. The methods developed by Leske and Buckley and Amores and colleagues are regionalized, but with a very limited coverage. Regionalized impacts in LCA are supported by geographic information systems (e.g. (Boulay et al. 2011b; Núñez et al. 2012; Núñez et al. 2009; Saad et al. 2013).

Midpoint - For water and soil salinization, two pathways modeling can be differentiated: on the one hand salinization induced by a LUC, which inventory flow is a land use transformation, and on the other hand irrigation-driven salinization, which inventory relies on a volume of water withdrawn (for saline intrusion) or released (waterlogging) and salts contents in water (salts released in soil and water bodies) (Cf. Figure 1). While LUC and saline intrusion result from a modified hydrologic (and thus salt) balance at regional or catchment scale, direct salts emissions through irrigation water occur at local scale. The system limit definition, and thus the consistency between LCI and LCIA modeling, is a crucial issue because salinization mechanisms occur at different spatial and temporal scales. In particular, the soil status is critical because it is both an environmental target and a part of the agricultural system. If soil is excluded from the technosphere, a salt emission is the salts content in irrigation water, while if soil is partially included in the system, a salt emission depends on soil conditions and practices. If soil is totally included in the system, no soil salinization would be accounted for and salt emissions would occur in the aquifer or river. Setting the system limit will determine which parameter has to be accounted for in the inventory or to be part of the CF.

We analyze in the following the modeling options for each AoP.

Ecosystems - Salinization affecting ecosystems could be modeled through aquatic ecotoxicity assessment for water salinization. Although salts are not classified as toxic substances, salts have toxic effects at high concentration. Amores et al. 2013 adopted this type of modeling in a specific context. Salts are not yet modeled in the USEtox model (Rosenbaum et al. 2008), but future developments of this tool could include freshwater ecotoxicity CFs for salts. Another future improvement of USEtox model is to develop regional versions because no spatial differentiation of location of the emission was considered so far (Henderson et al. 2011). Water salinization may also damage ecosystems through reduced water availability: a functionality-based approach could be adopted for this pathway. Similarly to water salinization, soil salinization affecting ecosystems could be modeled through terrestrial ecotoxicity assessment. The development of terrestrial ecotoxicology CFs is part of future developments of the USEtox model (Henderson et al. 2011).

Human health - The modeling of water salinization affecting human health could be approached by Boulay and colleagues framework. Total Dissolved Solids, Bicarbonate, Chloride, Chlorides/nitrites, Sodium and Sulfate in water are already parameters accounted for in this method to define the water categories for users (Boulay et al. 2011a). However, Boulay et al. (2011) method cannot be applied in its present form due to a scale modeling issue: the water salinized should be an inventory flow which is the result of a balance between water input and water output (with associated salinity increase). Boulay’s method can only be applied in the case of salinization of drainage water induced by irrigation, and if the soil is included in the technosphere; i.e the saline drained water is considered an “emission”. Soil salinization affects human health through reduced food supply. This is linked with the ecotoxicological effects of salts on crops, but also to soil physical degradation. Feitz et al. (2002) method is the mid-point step contributing to this pathway.

Resource - Water and soil salinization affecting water and soil resources are debatable pathways. In the case of permanently saline aquifers, we can consider that future generations will be deprived of water in that specific location. But as noticed previously, water quality alteration affecting resource is not considered in the water use impact framework. Regarding soil salinization, permanent degradation of soil or a loss of soil through erosion reduces soil availability as a future resource (Núñez et al. 2012). The UNEP SETAC Land use working group considers that a high salinity area in a very dry climate could be barren for an indefinite time period and corresponds to a permanent impact (Koellner et al. 2013b).

The land use impacts framework could cover soil salinization pathways, but within this framework, the inventory flux is a land use type, and not a salt release. According to JRC-IES 2011 “*soil salinization may be included in a completed/revised land use category”*. Soil salinization and waterlogging could be included in the global land use impact assessment on biodiversity and ecosystem services. Salinization, together with other mid-points, has an impact on the potential for biotic production (Brandão and Milà i Canals 2012). Also, salinization may impact freshwater regulation, erosion regulation and water purification, but this soil parameter is not accounted for in the method developed by Saad et al. (2013). To include salinization impact potential, a refinement of the land use types would be required to account for soil and agricultural practices and their effects on salinization. However, it is important to notice that accounting for salts emission through the land use framework reduces the frontier with the salt emission modeling framework, thus increasing double counting risks. Moreover, it is important to notice the link between land use and water use categories, especially for irrigated agriculture. Indeed, irrigation is part of the land use practices, and LUC can lead to changes in the water cycle (Koellner et al. 2013c).

5. Conclusion

Including salinization impacts in LCA is of high priority, especially for agricultural systems. Although the existing methods addressing salinization in LCA are important and relevant contributions, they are incomplete in terms of spatial and environmental mechanisms coverage. The modeling complexities lie in the inter-relations between salinization mechanisms, at both local and regional scales, and the status of soil and water in LCA which are both resources and living environments. An analysis of the modeling options in agreement with the LCA framework has been proposed in this paper. Much research effort is still required to include salinization impacts in a global, consistent and operational manner in LCA.

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The World Food LCA Database project: towards more accurate food datasets

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ABSTRACT

There is an increasing demand for LCA applied to the food and beverage sector. However, major limitations in doing LCA studies in this sector are currently the lack of inventory data and processes, and the absence of consistency among existing food datasets. There is a need to develop detailed, transparent, well-documented and reliable data in order to increase the accuracy and comparability of LCA in the food sector. This need is being addressed by the World Food LCA Database (WFLDB) project. The main aim of the WFLDB is to create a basis to assist companies and environmental authorities to assess and reduce ("eco-design") the impacts of food and beverage products, in initiatives like Environmental Product Declarations (EPD) or product labelling and also for academic research.

Keywords: Agriculture; database; environmental product declaration; inventory; LCI.

1. Introduction

Agricultural production and food processing contribute significantly to environmental impacts on global warming, eutrophication and acidification (Pardo and Zufia 2012; Ruviaro et al. 2012; Saarinen et al. 2012). The use of LCA for the assessment of these impacts is steadily increasing in the last decade (Notarnicola et al. 2012). However, major limitations to such assessments are the lack of reliable and consistent inventory data.

Existing libraries of LCA data on food are most often:

- Not transparent enough
- Incomplete: only few inventory flows are accounted for, which leads to an incomplete overview of the impacts of food products and misleading interpretations and conclusions
- Inconsistent among each other, due to different approaches and assumptions
- Outdated and consequently unreliable
- Not regionalized: country-specific data are rarely available

Therefore, it is critical to develop detailed, transparent, well-documented and reliable data to allow for more accurate and comparable LCA in the food sector.

In this context, Quantis and Agroscope launched early 2012 the World Food LCA Database (WFLDB) project which will be completed in 2015, in collaboration with, as of April 2014, ADEME, Bayer, Swiss Federal Office for the Environment, General Mills, Kraft Foods, Mars, Mondelēz International, Monsanto, Nestlé, Syngenta and Yara.

The main aim of the WFLDB is to create a basis to assist companies and environmental authorities to assess and reduce ("eco-design") the impacts of food and beverage products, in initiatives like Environmental Product Declarations (EPD) or product labelling and also for academic research.

2. Methods

A new set of food inventory data is being developed from existing LCA studies on food products (project partners' past LCAs, Agroscope and Quantis existing databases), literature reviews, statistical databases of governments and international organizations (such as the Food and Agriculture Organization of the United Nations), environmental reports from private companies, technical reports on food and agriculture, information on production processes provided by the project partners as well as primary data. Background datasets from the ecoinvent database are being used as a basis and compatibility with ecoinvent will be ensured.

The developed datasets include, when relevant, different production schemes (such as conventional, integrated or organic production), regional specificities and deforestation impact.

To guarantee its transparency, the inventory database is fully documented, unit processes are visible (except for confidential data) and all sources are referenced. The end-user will be able to differentiate among different stages of the process (e.g. agricultural production vs. food product manufacturing) and to identify the main impact contributors for each dataset (e.g. pesticides, fertilizer use, etc.).

Datasets created within the project will initially be solely available to the project partners and they will become public through their integration in ecoinvent.

The scientific modelling principles of the WFLDB are at a first instance based on:

1. ISO standards14040 and 14044 (ISO, 2006a; ISO 2006b)
2. ecoinvent quality guidelines (ecoinvent, 2013; Weidema et al, 2013)
3. ILCD guidelines (JRC, 2010)

The project managers collect data and define the methodological guidelines for modelling the WFLDB datasets. These rules are based on the above-mentioned documents and on other existing guidelines for modelling agricultural processes. By doing this, it is ensured that all datasets within the WFLDB are modelled according to internationally accepted standards and are fully consistent with each other. Furthermore, the project managers follow the developments within other international initiatives and organizations such as The Sustainability Consortium (TSC), the Food and Agriculture Organization of the United Nations (FAO), the Sustainable Agriculture Initiative Platform (SAI), the EU Food SCP Roundtable and the EU Product Environmental Footprint. Scientific guidelines used in other database initiatives such as Agri-BALYSE (Van der Werf et al., 2010) and ACYVIA (Bosque et al, 2012) are also considered for the definition of the WFLDB modelling principles (developments are considered as they occur).

The methodological guidelines of the World Food LCA Database will be reviewed by a selected panel of external and independent experts and will be published in 2014.

The aim is to be as compliant as possible with the developments occurring within the above mentioned initiatives and organizations. Compliance with other initiatives is assured by the project advisory board, which has a consultative role and is constituted of members of non-governmental and research organizations.

The WFLDB datasets will be delivered in the most widely used data exchange formats for LCA software (i.e. ecospold v1, ecospold v2, SimaPro-CSV, Quantis SUITE 2.0-excel). This will enable using the datasets in common LCA software: SimaPro, GaBi, OpenLCA and Quantis SUITE 2.0. The datasets are released in three phases, the first release was in summer 2013. Two years after their release to the project partners, datasets will be submitted to the ecoinvent Centre for their integration in the ecoinvent database.

3. Conclusion

The WFLDB will be a comprehensive LCA food database providing detailed LCI data of high scientific quality, reliability and transparency, while being in line with other database developments such as ecoinvent v3.

The database will provide a large number of new food-related inventory datasets with a focus on different production schemes and regional specificities.

As an illustrative example, Figure 1 shows the preliminary results for carbon footprint of some processes related to coffee, from processes fairly upstream the life cycle of coffee (e.g., green coffee production) until fairly downstream the life cycle of coffee (e.g., decaffeinated instant coffee), for different technologies (e.g., spray and freeze drying). Furthermore, specific gate-to-gate steps are also modelled (such as roasting and grinding) and could be combined with customized processes (for example regional coffee production that would be modelled with primary data). This gives high flexibility to the final user of the WFLDB.

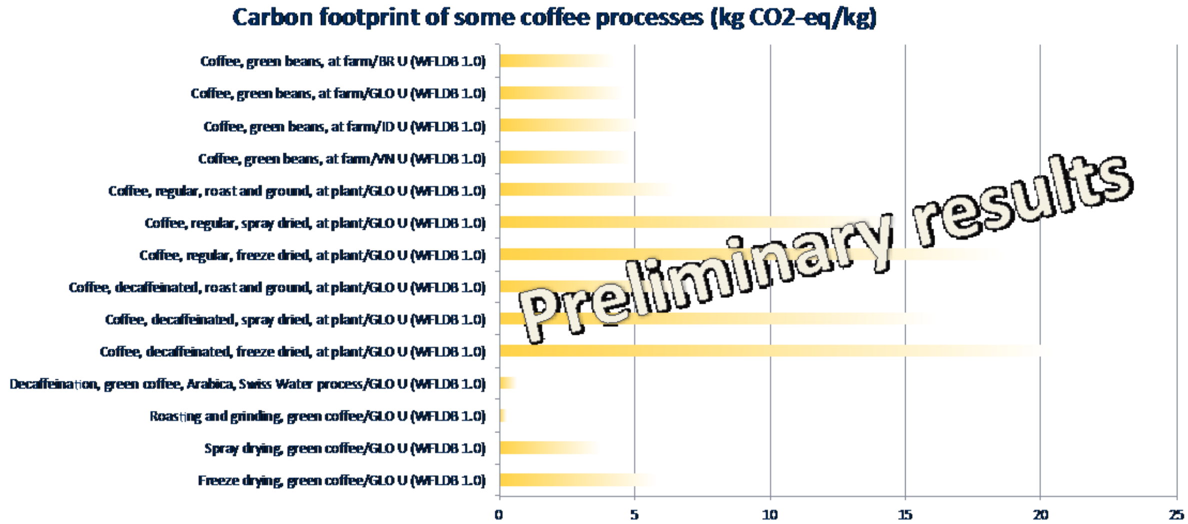

Figure 1. Preliminary results for carbon footprint of some processes related to coffee

Learning obtained in the last 2 years data collection as well as recommendations for any practitioner and company encountering challenges with food data collection will be presented.

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The EU Organisation Environmental Footprint applied to the Retail sector

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ABSTRACT

On April 9th 2013, the European Commission published a communication to the European Parliament and the Council to Build a Single Market for Green Products to facilitating better information on the environmental performance of products and organisations. An open call for volunteers was announced by the European Commission for the Product Environmental Footprint (PEF) and the Organisation Environmental Footprint (OEF), inviting companies, industrial and stakeholder organisations in the EU to participate in the development of product-group specific and sector-specific rules. A group of public and private organisations has been selected by the European Commission to develop the guidance for surveying and reporting environmental impacts in the European retail sector. The Technical Secretariat which is responsible for developing the OEF sector rules in two years (official launch in November 2013), is composed by four retailers: Carrefour SA, Colruyt Group, Oxylane Group (Decathlon) and Picard; three public agencies: Environment Agency Austria (EAA), French Environment and Energy Management Agency (ADEME) and Italian National agency for New Technologies, Energy and Sustainable Economic Development (ENEA); one non-governmental organization: Global 2000; one association PERIFEM; and one Life Cycle Assessment (LCA) consultant: Quantis. The project, challenges and preliminary results and benefits of this pilot test will be presented highlighting feedback in reference to specific modelling issues related to the application of LCA to a sector as vast as the retail sector such as defining system boundaries (e.g., direct, as well as upstream and downstream indirect contributions) and choosing life cycle impact assessment methods (e.g. which indicators are relevant, which weighting scheme to use). These points also include the issue pertaining to consistency with the product approach for a sector as interdisciplinary as the retail sector.

Keywords: Organisation Environmental Footprint (OEF), Sector rules (SR), Product category rules (PCR), European Commission (EC)

1. Introduction

On April 9th 2013, the European Commission (EC) published the following: "Communication from the Commission to the European Parliament and the Council: Building the Single Market for Green Products, facilitating better information on the environmental performance of products and organisations" [1].

An open call for volunteers was announced by the EC for the Product Environmental Footprint (PEF) and the Organisation Environmental Footprint (OEF), inviting companies, industrial and stakeholder organisations in the EU to participate in the development of product-group specific and sector-specific rules.

A group of public and private organisations has been selected by the ECn to develop the guidance for surveying and reporting environmental impacts in the European retail sector [2]. The Technical Secretariat which is responsible for developing the OEF sector rules in two years (official launch in November 2013), is composed by three retailers: Carrefour SA, Colruyt Group and Oxylane Group (Decathlon); three public agencies: Environment Agency Austria (EAA), French Environment and Energy Management Agency (ADEME) and Italian National agency for New Technologies, Energy and Sustainable Economic Development (ENEA); one NGO: Global 2000; and one LCA consultant: Quantis.

2. Methods

The approach used to develop the OEFSR will encompass the four following principles: (1) life cycle-based approach; (2) multi-criteria; (3) physically realistic modelling; (4) reproducibility /comparability.

This pilot will test how approaches such as Product Environmental Footprint Category Rules (PEFCRs) and the "Chain OEF" (approach covering the indirect/upstream part of the value chain) will interact with or benefit the proposed OEFSR. The Chain-OEF aims primarily to allow the assessment of the environmental footprint based on the product portfolio of retailers, produced or not by them, using a cascade system; and to involve progressively more and more companies in the supply chain, enhancing primary data collection and building transparent partnerships. This sub-pilot strives to develop a cost-efficient approach to analyse, link and reduce the impact of each player in the value chain.

3. Results

Table 1 presents the preliminary results in terms of scope of the OEFSR.

Application	Use of an OEFSR		
	Optional "May"	*Recommended "Should"*	*Mandatory "Shall"*
In-house: improvement of the organisation's environmental performance	✓		
Reporting without comparisons or comparative assertions		✓	
Reporting with comparisons or comparative assertions			✓
Any OEF study declared to be in compliance with the OEF Guide	✓		

Table 1: Preliminary scope of the OEFSR (scenarios that does and does not necessitate the use of OEFSRs)

Building upon OEF studies already carried out, several aspects are identified as being challenging, including the definition of (i) the representative organisation model, (ii) benchmark and classes of environmental performance, or (iii) a weighting scheme different from the one proposed by the EC.

Figure 1 presents the system boundary of the OEFSR in general.

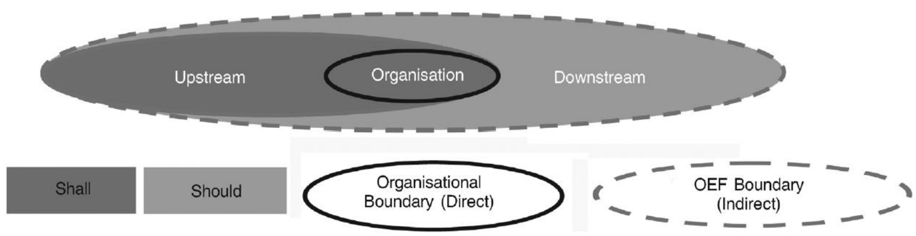

Figure 1: System boundary of the OEFSR in general

Figure 2 presents the system boundary of the OEFSR adapted to the retail sector (initial proposal for discussion).

Figure 2: System boundary of the OEFSR adapted to the retail sector

4. Conclusion

The project, challenges, preliminary results and benefits of this pilot test will be presented highlighting feedback in reference to specific modelling issues related to the application of LCA to a sector as vast as the retail sector such as defining system boundaries (e.g., direct, as well as upstream and downstream indirect contributions) and choosing life cycle impact assessment methods (e.g. which indicators are relevant, which weighting scheme to use). These points also include the issue pertaining to consistency with the product approach for a sector as interdisciplinary as the retail sector.

Numerical results of overall retailer OEF as well as proposed rules to assess the OEF of retailers will be available for public consultation for four weeks starting around December 2014. Final rules and generic results should be available by February 2015. Preliminary results and rules will be presented at LCAFood2014 (but not presented here as they are being calculated until end of September) to invite interested experts to engage in the public consultation that will occur at the end of the year and allow them to start considering strength and weaknesses of the proposed approach as well as technical, political, reputational and financial implication for the different economical sectors.

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Grass as a C booster for manure-biogas in Estonia: a consequential LCA

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ABSTRACT

The aim of this study was to assess the environmental consequences of using grass (from both unused and cultivated boreal grasslands) as a co-substrate to dairy cow manure for biogas production. Environmental impact categories assessed were global warming, acidification and nutrient enrichment (distinguishing between N and P). Scenarios studied were: traditional management of dairy cow manure, mono-digestion of manure, manure co-digestion with reed canary grass and manure co-digestion with residual grass from semi-natural grasslands. The latter scenario showed the best environmental performance for the global warming category, for other categories it did not show clear benefits. Using reed canary grass specially produced for biogas purpose resulted in a climate change impact just as big as the reference manure management, mainly as a result of indirect land use changes. Increased impacts also occurred in the acidification and eutrophication (N) categories for the reed canary grass scenario, reflecting the impacts of the cultivation process. The main conclusion was that future strategies for manure-biogas production in Estonia should not rely upon land-dependent biomass, even if the availability of arable land in Estonia is, under current conditions, not considered to be an issue.

Keywords: anaerobic digestion, land use changes, dairy manure, reed canary grass, natural grass

1. Introduction

Biogas production from manure has a good potential to simultaneously produce a renewable and flexible energy carrier, while reducing the environmental impacts of manure management (mainly due to the reduced emissions from raw manure storage) and recycling biomass macronutrients (as well as the slowly degradable carbon) (Hamelin 2013). Although the energy produced from manure-biogas in the European Union (EU) is currently far below its full potential (Hamelin et al. 2014), a drastic increase of biogas production is nevertheless planned in the EU (Beurskens and Hekkenberg 2011), as well as in Estonia (Melts et al. 2013).

However, due to the too low carbon (C) and carbon-to-nitrogen (C/N) content of animal manure, it is usual practice to supplement manure with C-rich co-substrates for anaerobic digestion. Grass, especially reed canary grass, has been considered to have great potential for biogas production mainly due to its relatively high yield and the fact that arable land resource is available in Estonia (Ministry of Economic Affairs and Communications 2010; Värnik et al. 2011).

Two grass options were considered in this study: i) reed canary grass (this being one of the dedicated energy crops suggested to grow in Nordic countries) and ii) the residual grass from semi-natural grasslands, which is clearly underused currently but has a considerable biogas potential (Melts et al. 2013).

The goal of this consequential life cycle assessment (LCA) study was to quantify the environmental consequences of implementing, in Estonia, a manure-biogas strategy relying on grass as a co-substrate (options i) and ii), as opposed to managing manure conventionally and not harvesting the grass from semi-natural areas, nor producing energy grass. The focus is on dairy cow manure, this being presenting the highest share from all manure types in Estonia (Luostarinen 2013).

2. Methods

2.1. LCA approach

The life cycle impact assessment methodology used for this study was the EDIP2003 method described in Hauschild and Potting (2005) and the functional unit upon which all input and output flows were expressed was ”the management of 1 tonne of dairy cow manure ex-animal (i.e. the manure as freshly excreted by the animals)”. Four impact categories were considered: global warming, acidification and nutrient enrichment (distinguishing between N and P). Background data were based on Ecoinvent v.2.2 database (Frischknecht and Rebitzer 2005). Foreground data were mainly based on the Estonian situation, partly combined with Danish data.

The life cycle inventory and process flows are detailed in Pehme (2013), and the assessment was facilitated by the software SimaPro 7.3.2. The geographical scope was considered to be Estonia, i.e., inventory data for biomass composition, technologies and emissions were specific to the Estonian/Baltic conditions.

In this study, biogenic carbon flows (both removals from atmosphere by plants and also emissions) were fully accounted for each process.

2.2. System boundaries and description of scenarios

Four different scenarios are considered in this study: one reference scenario (conventional management of dairy cow manure) and three biogas alternatives (mono-digestion; co-digestion with energy grass; and co-digestion with grass from semi-natural areas).

In the reference scenario, dairy cow manure is handled as slurry after excretion, pumped towards outdoor storage at least once per day, stored outside in a concrete slurry tank covered by a naturally-forming crust layer and applied to fields when suitable. The life cycle inventory of the reference manure management is detailed in Hamelin et al. (2013), including details on the manure composition. The process flow diagram of this scenario is presented in Figure 1, reflecting the mass changes of manure due to emission losses and water addition in-house and at the outdoor storage through precipitation. The dry matter (DM) content of dairy manure ex-housing (i.e. as it leaves the housing unit) considered in this study is 11.5%, the volatile solids (VS) representing 82.0% of the DM (Hamelin et al., 2013).

In the biogas scenarios, manure is instead collected from the animal houses and used in biogas plants, digested in a mesophilic 2-steps digestion process. The biogas was considered to be used for combined heat and power production (CHP). The marginal energy sources displaced by the biogas were natural gas for the heat and oil shale for the electricity. The digestate was assumed to be stored and used as a fertilizer, displacing the marginal mineral nitrogen, phosphorus and potassium fertilizers for Europe (respectively taken as calcium ammonium nitrate, diammonium phosphate and potassium chloride; Hamelin 2013). The rationale behind this fertilizer substitution is, based on the Estonian context, that if the farmer would not have had the manure or digestate, the farmer would have applied mineral fertilizers up to the crop needs and national regulations. Yet, this does not mean that 100% of the N, P and K applied with the raw manure (reference scenario) and digestate (biogas scenarios) correspond to avoided mineral fertilizer; only the portion available to plants was considered to avoid the production and use of mineral N, P and K. The full calculation of avoided fertilizers is detailed in Pehme (2013).

For the 2 scenarios involving co-substrates, it was considered, based on Hamelin et al. (2011), that these were added to manure in order to get an input mixture with a DM content of 10% after the first digestion step. Fugitive CH_4 losses from the anaerobic digestion process were taken as 1% of the overall CH_4 produced, based on Hamelin et al. (2014) and assuming the implementation of state-of-the-art biogas technologies. Further life cycle inventory data, details and mass balances for all biogas scenarios are detailed in Pehme (2013).

The co-digestion with energy grass scenario, here referred to as the “Reed canary grass (RCG) scenario”, is based on co-digestion of dairy cow slurry and reed canary grass silage. RCG is here produced specially for biogas purpose, fertilized and harvested twice per year. The average grass yield for a 15 years plantation is considered as 8.23 t DM/year. RCG production data were based on Värnik et al. (2011). The production of RCG is considered to displace the use of land for cultivating barley, thus this barley cannot be produced on the same land and has to be produced somewhere else, thus involving land use changes emissions (expansion and intensification). Different life cycle assessments have identified spring barley as the marginal crop displaced by an increased demand for other crops (e.g. Hamelin et al. 2012; De Vries et al. 2012). Barley is mostly produced in areas with lower soil quality and its gross-margin value is lower compared to other crops. Thus, production of energy grass instead of barley has been presented as an attractive choice for Estonian farmers from the economic and agronomic point of view (Värnik et al. 2011). In this study, indirect land use changes (ILUC) emissions of 357 t CO_2 eq. per ha of barley displaced were considered on the basis of (Hamelin et al. 2014), which corresponds to 18 t CO_2 eq. ha per year (20 years annualization). Hamelin et al. (2014) derived that estimate from the results of Kløverpris (2008) for a marginal increase in wheat consumption in Denmark. Process flows for the RCG scenario are illustrated in Figure 2.

The co-digestion scenario involving natural grass (NG) is based on co-digestion of dairy cow slurry and natural grass silage. Grass is collected once per year from semi-natural grassland (alluvial meadows) where no soil cultivation is practiced and no agrochemical inputs are used. The grass yield is assumed to be 5.5 t DM/ha according to Melts et al. (2013). Currently there is no use for most of the biomass from those areas, so harvesting the grass prevents it to decompose and cause CO_2 emissions to the atmosphere. This (avoided) decay process was modeled following a first order ($C_t=C_oe^{-kt}$) decay, assuming that 100% of the above-ground biomass is transferred to the soil, and using the decay rates of Freschet et al. (2013). On the basis of this, it was considered that 100% of the C in the above-ground grass biomass would have been emitted as CO_2, if the grass would not have been harvested. In this system, this translates to avoided CO_2 emissions of 9.5 t CO_2 eq. ha per year (20 years annualization), considering a C content in the grass biomass of 0.47 kg C kg^{-1} DM. Of course, as the grass C ends up to be emitted through the biogas scenario (among others in the biogas and through the application of the digestate), this credit is, at the end, essentially counterbalanced. Production, cutting, chopping and transport of grass are accounted in the analyses for both grass co-digestion scenarios.

More straightforward, the mono-digestion scenario is based on the anaerobic digestion of dairy cow slurry ex-housing, this being the only substrate. The digestate then undergoes the same processes as for the other biogas scenarios.

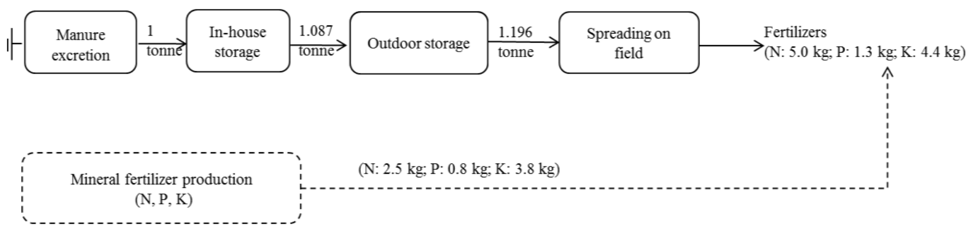

Figure 1. Process flow diagram for the reference manure management scenario per 1 tonne of manure ex-animal

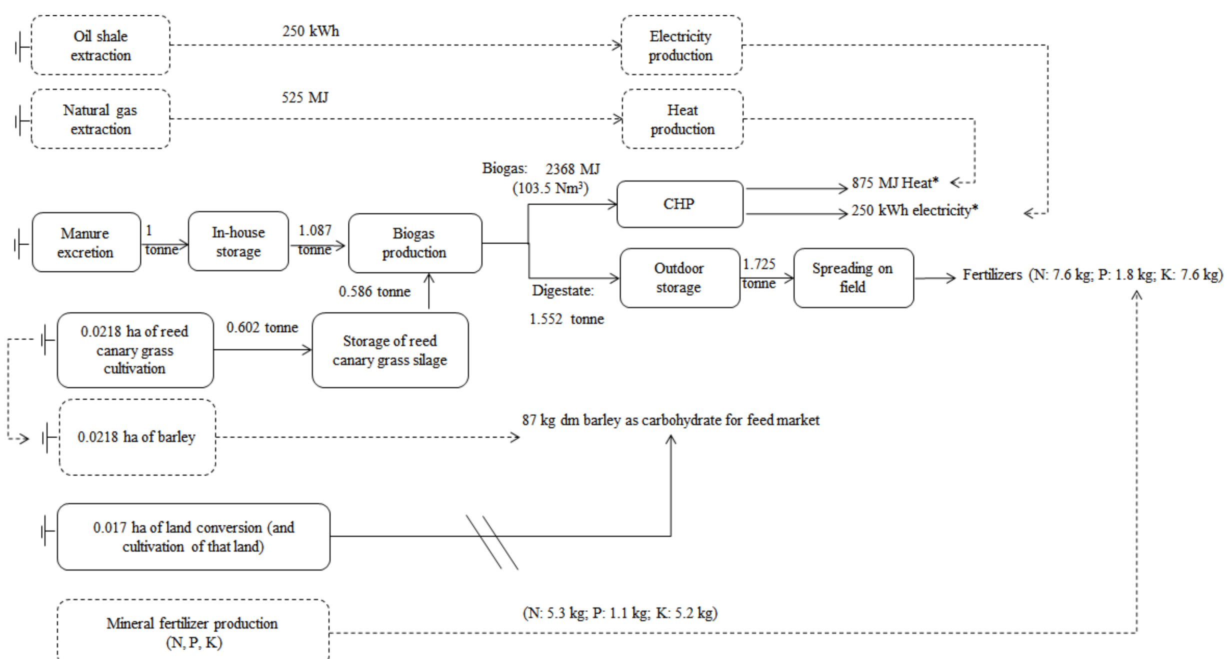

Figure 2. Process flow diagram for the reed canary grass scenario per 1 tonne of manure ex-animal

3. Results

The natural grass scenario had the best reduction potential for the global warming impact category (Table 1). The reed canary grass scenario showed equal result compared to reference scenario; the main reason for this was indirect land use changes. Mono-digestion displayed a good potential to reduce the global warming impact, but its energy production is significantly lower.

Most of the induced greenhouse gas (GHG) emissions originated from field application of manure and digestates, and from the burning of the biogas in the biogas engine prior to CHP. For all scenarios, the main reduction of GHG emissions was caused by the avoided oil shale-based electricity.

For the rest of the impact categories, natural grass did not show clear environmental benefits in comparison to the reference scenario (conventional manure management without biogas) (Table 1).

Table 1. LCA results, per 1 tonne of dairy cow manure ex-animal.

Impact category	**Reference manure**	**Mono-digestion**	**Co-digestion with RCG**	**Co-digestion with NG**
Global warming, kg CO_2 eq.	314	155	314	-207
Acidification, m^2 "unprotected ecosystems eq." (UES)	43	37	60	48
Aquatic Eutrophication, N eq.	0.40	0.28	0.89	0.38
Aquatic Eutrophication, P eq.	-0.02	-0.02	0.01	0.04
Grass input to digester per FU, tonne	-	-	0.586	0.407
Energy produced per FU, MJ	-	860	2368	1916

The highest contributions to the acidification category were caused by the field application and outdoor storage for all scenarios, reflecting essentially the losses of nitrogen as ammonia. For the RCG scenario, additional emissions were caused by the grass cultivation process. For the N and P eutrophication, the main contributions came from field application and the main emission reductions originated from the avoided mineral fertilizers production and application.

4. Discussion

The results of this study highlighted the important potential environmental impacts related to the use of land-dependent biomass, when implementing a national renewable energy strategy (in this case based on manure-biogas). In this study, co-digesting dairy cow manure with dedicated RCG resulted in an overall worse environmental performance than not producing biogas at all (i.e. the reference scenario where heat and power are based on fossil fuels and manure is managed conventionally). Similar conclusions are presented in some studies (De Vries et al. 2012; Hamelin et al. 2014), but very often bioenergy studies exclude the land use change impacts. In the Estonian context, in can indeed be debated whether it is reasonable to consider that cultivating RCG would lead to the displacement of barley, given the great availability of uncultivated land. However, if the Estonian stakeholders are really serious about a manure-biogas strategy relying upon the supply of dedicated energy grass, it seems reasonable to assume that the frontier between the availability of arable land (supply) and the demand for it will be reached. This situation is exactly what this LCA endeavored to model, in the aim of preventing eventual misleading decisions.

Residual biomass from nature conservation areas as it is illustrated by the natural grass scenario results of this study should be preferred to cultural grass to achieve the target for increased biogas production reflected in the National Renewable Energy Action Plan of Estonia 2020 (Ministry of Economic Affairs and Communications 2010). Natural grass shows great reduction potential especially in global warming category, but there are technological issues to solve connected to the access to harvesting mainly due to the seasonal flooding (Heinsoo et al. 2010). Managing natural areas has also other benefit not reflected in the LCA results– it would ensure to maintain their high biodiversity value. Grass yield from floodplain meadows in Estonia have been estimated to

113,349 tonne of DM (Heinsoo et al. 2010). If half of the biomass is considered to stay unused currently and would be used for anaerobic digestion in mixture with manure as presented in this study, it would result in a biogas amount of 11.4 x 10^{-6} Nm^3, which would correspond to a greenhouse gas emissions reduction of approximately 28,000 t CO_2 eq.

However, this study did not reflect the practical aspects of using grass for anaerobic digestion. Economic aspects of grass collection, feasibility of harvesting, possible impacts on the digestion process (e.g. corrosion) need further investigation.

It can also be debated whether the alternative use of the natural grass would, on a long-term perspective, be to be left on land. If, for example, this grass has a high protein value and could become competitive enough to be used for animal feed, then a protein feedstuff is displaced. In such case, it is likely that no environmental benefits would be obtained from using the grass as a co-substrate to manure-biogas, as e.g. shown in De Vries et al. (2012) for agro-industrial residues with high protein value.

5. Conclusion

Manure-biogas does, for the Estonian context, lead to significant benefits. In this study, manure-biogas strategies were shown to yield overall environmental benefits if based on grass co-substrates from natural areas, or if simply based on mono-digestion. Yet, a strategy relying on the use of dedicated reed canary grass was shown to lead to an overall worse environmental performance than not producing biogas at all. This was essentially due to the impacts of the cultivation process itself, as well as to the cascading effects involved when considering the use that the land would have otherwise had, if not used for dedicated energy grass cultivation.

6. Acknowledgements

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Modeling fuel use for specific farm machinery and operations of wheat production

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ABSTRACT

This paper presents the approach used to develop a model predicting fuel consumption according to the different agricultural practices and the results obtained for wheat production. Results showed that the largest variations were caused, not by production systems, but by soil preparation and fertilization operations. Average fuel consumption was 55.6 L/ha regardless of the production system, the soil preparation and fertilization strategies and the type of wheat. On average, soil preparation consumes 32% of the fuel, but it can reach more than 50% if conventional tillage is used. Grouped by tillage system, instead of production systems, no-till plots consume 39.6 L/ha, reduce tillage 55.6 L/ha and conventional 76.0 L/ha. Fuel consumption due to fertilization varies according to the type of fertilizer used. The amount of fuel used seems to be related to the concentration of nitrogen in the fertilizer. Winter wheat production used less fuel than spring wheat.

Keywords: life cycle assessment, wheat production, greenhouse gas, environmental impact, fuel consumption

1. Introduction

Life cycle assessment (LCA) performed on agricultural products generally use average values of fuel, fertilizer and pesticide consumption per hectare of land. These average values are close to reality when the LCA results are used to inform consumers about the environmental impact of products like milk or flour that have been produced on more than one farm using different agricultural practices. Such values are, however, less relevant for agricultural producers who want to improve the environmental impact of their farm.

Several factors may influence fuel consumption: production systems, soil and fertilizer management and technical factors such as the width and weight of equipment, as well as the power of the tractor used during farm operations.

A model predicting fuel used for specific farm machinery and operation was developed while performing a “cradle-to-farm gate” LCA of wheat production. The aim of this LCA was to assess the environmental impact of four production systems (intensive, conventional, integrated and organic) for spring and winter wheat.

This paper focuses on the approach used to develop the model and the results obtained for fuel consumption according to the different agricultural systems.

2. Methods

2.1. Inventory data

Four production systems for spring and winter wheat were considered in this study. Difference in production systems were observed in the management of fertilizer and pesticide use. Organic producers had to be accredited by a certified organic body. Integrated producers were obliged to follow a book of specifications provided by the grain buyer. They were not restricted with the use of mineral fertilizer alongside manure, but they were encouraged to respect a fertilization management plan. They were only allowed to use herbicide before seeding, but not during the growing season. Conventional and intensive producers were not restricted in the use of fertilizers or pesticides. Intensive producers tended to use more pesticides in combination with a growth regulator. Regardless of the production system, soil management practices were not limited.

Data used in the project come from plots established on farms that were accustomed to sell their wheat to a grain buyer. Producers provided data for soil cultivation, like tillage, seedbed preparation, seeding, fertilizers and pesticides application, harvesting and on-farm transportation. They were asked to provide, for every operation, the main characteristics of the implement (e.g., width and weight), the tractor used (e.g., type and power) and the working conditions (e.g., yield, soil texture, travel speed and working depth). Producers with information such as fuel consumption and field capacity, recorded by the GPS of their tractor or combine were invited to provide them. These values were thereafter used to validate the model.

Based on the list of equipment provided by the producers, a data base of equipment has been developed alongside the model. This database includes the following equipment:

- Chisel
- Combine (auger or draper head, rigid or flexible)
- Disk gang
- Disk harrow
- Field cultivator
- Grain drill (rows or no-till)
- Grain wagon
- Loader
- Manure pump
- Manure spreader (solid and liquid)
- Mounted fertilizer spreader
- Plow (semi-mounted and mounted)
- Rod weeder
- Roller packer
- Sweep plow
- Pesticides sprayer (trailed)

2.2. Fuel consumption

The methodology used for the calculation of fuel consumption was based on the equations presented in the following two standards produced by the American Society of Agricultural and Biological Engineers: Agricultural Machinery Management Data (ASAE D497.7 Mar2011; ASABE, 2011) and Agricultural Machinery Management (ASAE EP496.3 FEB2006; ASABE, 2006). The methodology estimates fuel consumption of an operation using the equipment characteristics (e.g., width, mass and tillage depth) as well as the characteristics of the tractor used (e.g., mass and travel speed).

The methodology has been simplified according to the available data. A summary of the main calculation is presented here. The calculation of fuel consumption can be summarized by the following equation:

$$Q_{fuel} = \frac{P_{total} \times Q_{sfc}}{C_{field}} \qquad \text{Eq. 1}$$

Where Q_{fuel} is the fuel consumption (L/ha), P_{total} is total power requirement for an operation (kW), Q_{sfc} is specific fuel consumption (L/kWh) and C_{field} is field capacity (ha/h).

Total power (P_{total}) requirement for operating implements is the sum of implement power components (Eq. 2).

$$P_{total} = \frac{P_{db}}{E_m E_t} + P_{pto} + P_{hyd} + P_{el} \qquad \text{Eq. 2}$$

Drawbar power (P_{db}) is power developed by the drive wheels or tracks and transmitted through the hitch or drawbar to move an implement through or over the crop or soil (ASABE, 2006). It is primarily a function of the width of the implement and the speed at which it is pulled. It also depends upon soil texture, depth and geometry of the tool. Drawbar power is the sum of motion resistance and soil and crop resistance multiply by the travel speed. Mechanical efficiency (E_m) and tractive efficiency (E_t) are coefficients specific to tractor characteristics.

Power-takeoff power (P_{pto}) is the power required by the implement from the PTO shaft of the tractor or engine and it is a function of the width of the implement and the material feed rate or yield.

Hydraulic power (P_{hyd}) is the fluid power required by the implement from the hydraulic system of the tractor or engine. Electric power (P_{el}) is required to operate components of implements. Hydraulic and electric power are difficult to assess and were not considered in the survey, they were replaced by a factor ($F_{Phyd\ +\ Pel}$) representing an additional 25% of the power requirement.

Total engine power must be greater than the total implement power required. Additional power is required to accelerate and overcome changes in topography, soil and crop conditions (ASABE, 2006). An additional 20% of the power requirement has been allowed for reserve power (F_{Pres}).

For the purpose of the project, Equation 2 has been simplified and can be presented as:

$$P_{total} = \left(\frac{P_{db}}{E_m E_t} + P_{pto}\right) \times F_{P_{hyd}\,+\,P_{el}} \times F_{P_{res}} \qquad \text{Eq. 3}$$

Specific fuel consumption (Q_{sfc}) is the fuel requirement based on the actual power required. For a particular operation it can be found by multiplying specific fuel consumption volume by current power delivery. It takes into account the ratio of equivalent PTO power required by the current operation to rated power available and the ratio of partial throttle engine speed to full throttle engine speed.

Effective field capacity (C_{field}) is a function of field speed, machine working width, field efficiency and unit yield of the field.

3. Results

Average fuel consumption was 55.6 L/ha with a standard deviation of 21 L/ha for the 80 plots regardless of the production system, the soil preparation and fertilization strategies and the type of wheat. There was lot of variation in the results, the minimum value was 27.5 L/ha and the maximum value was 108.0 L/ha. Table 1 presents the average fuel used per operation according to production and tillage systems. Results showed that the largest variations in fuel consumption were caused, not by production systems, but by soil preparation and fertilization operations.

Table 1. Average fuel used for wheat cultivation per operation in function of production and tillage systems

Production and tillage systems	n	Yield (t/ha)	Soil preparation (l/ha)	Fertilization (l/ha)	Pulverization (l/ha)	Seeding (l/ha)	Harvesting (l/ha)	Transportation (l/ha)	Total (l/ha)
Organic									
No-till	3	4.2	-	4.7	-	11.2	9.4	4.3	29.7
Reduce	9	2.3	33.3	4.4	-	4.5	12.1	2.3	56.6
Conventional	3	3.0	34.0	5.6	-	7.6	11.9	3.1	62.2
Average		2.8	26.8	4.7	-	6.5	11.5	2.9	52.3
Integrated									
No-till	8	3.8	-	4.1	1.7	14.9	13.5	3.9	38.1
Reduce	6	3.6	12.8	13.2	0.6	8.6	13.8	3.7	52.6
Conventional	6	3.5	41.0	6.7	0.9	9.2	16.1	3.6	77.5
Average		3.6	16.3	7.6	1.1	11.3	14.4	3.7	54.5
Conventional									
No-till	8	4.4	-	4.3	2.4	14.9	13.5	4.5	39.6
Reduce	5	4.0	14.2	15.5	2.3	9.1	14.1	4.1	59.4
Conventional	6	4.2	41.0	6.7	2.5	9.2	16.1	4.3	79.8
Average		4.2	16.9	8.0	2.4	11.6	14.5	4.3	57.8
Intensive									
No-till	8	5.0	-	4.7	4.9	14.9	13.5	5.2	43.2
Reduce	12	4.4	10.7	13.5	5.0	7.9	13.3	4.5	54.9
Conventional	6	4.9	37.4	7.5	3.9	8.7	15.0	5.0	77.5
Average		4.7	13.7	9.4	4.7	10.2	13.7	4.8	56.7
Average		4.0	17.6	7.7	2.4	10.1	13.7	4.1	55.6

On average, soil preparation consumes 32% of the fuel, or 17.6 L/ha, but it can reach more than 50% or 41 L/ha. Grouped by tillage system instead of production systems, no-till plots consume 39.6 L/ha, reduce tillage 55.6 L/ha and conventional 76.0 L/ha. Conventional tillage system used roughly the same amount of fuel regardless of the production system. Soil preparation consisted, on average, of 2.3 operations, using a plow and one or two passes of field cultivator. For integrated, conventional and intensive production systems, reduce tillage consumed approximately one third of the fuel used by conventional tillage. An average of 1.6 operations was done by the producers combining a chisel, a field cultivator, a disk harrow or a rod weeder. Soil preparation was performed with an average of 3.8 operations for organic producers using reduce tillage. They consumed approximately the same quantity of fuel than those using conventional tillage (33.3 L/ha vs 34.0 L/ha). The exact reason for all these operations was not specified, but it may be due to a related operation such as weed control.

Fuel consumption due to fertilization varies according to the type of fertilizer used. Fertilization with ammonium nitrate averaged 1.8 L/ha or 0.9 l/ha per pass. Spreading of liquid cow manure averaged 14.7 L/ha while mixing and loading accounted for 16.4 L/ha for a total of 31.1 L/ha. Spreading of liquid pig manure averaged 7.7 L/ha while loading accounted for 4.8 L/ha for a total of 12.5 L/ha. Spreading of solid poultry manure averaged 3.6 L/ha while loading accounted for 2.1 L/ha for a total of 5.7 L/ha. The amount of fuel used seems to be related to the concentration of nitrogen in the fertilizer (Figure 1).

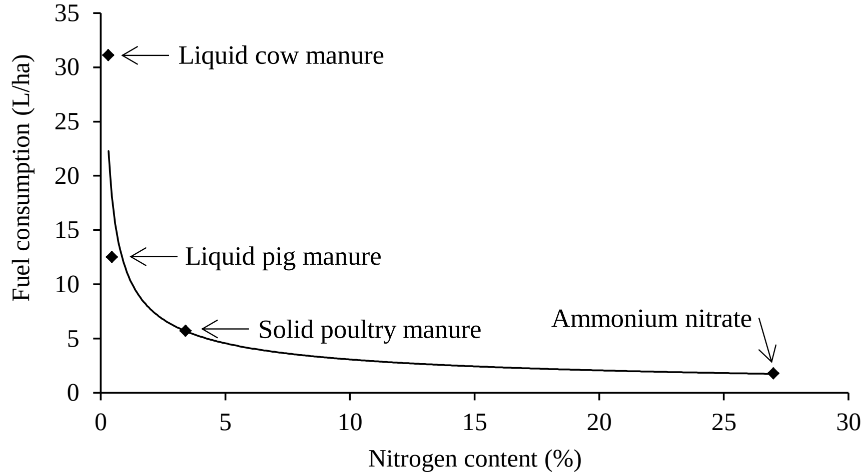

Figure 1. Fuel consumption of fertilization operations according to the concentration of nitrogen in the fertilizer

Winter wheat production used less fuel than spring wheat. Major differences were observed because more producers used no-till and fertilized with manure. Average fuel consumption for winter wheat was 53.6 L/ha with no-till representing 60% of the plots. Fertilization consume 11.4 L/ha and seeding 11.8 L/ha. Average spring wheat fuel consumption was 57.5 L/ha with no-till representing only 3 plots out of 40. Fertilization consume 4.1 L/ha and seeding 8.4 L/ha.

Pesticides pulverization consumed less fuel than any other operation. Integrated production system consumed 1.1 L/ha, conventional 2.4 L/ha and intensive 4.7 L/ha.

No-till seeding consumed approximately 5 litres more per hectare than seeding done after reduce or conventional tillage. Organic farming consumed less fuel for seeding because, for most of the plots, the operation was done using a tractor smaller than 100 kW. The same conclusion can be drawn for harvesting; the combines used in the organic plots were smaller.

4. Discussion

Sensitivity analysis should be performed to investigate the impact of farm equipment use. Efforts to reduce fuel use during farm operations would contribute to reduce greenhouse gas emissions, but this parameter has little influence on the whole farm emissions. Fuel consumption of 55 L/ha is equivalent to approximately 150 kg CO_2e/ha, which is not an important parameter compare to the emissions produced on a hectare of land. Furthermore, results observed in this study showed a variation corresponding roughly to 20 L/ha or 50 kg CO_2e/ha. In another vein, at the end of the year, an economy of 20 L/ha can represents a substantial amount of money for a producer.

If the objective is to reduce the environmental impact, the choice to use a machine or another must take into account all the operations of the farm. For example, the use of manure instead of mineral fertilizer increase greenhouse gas produced by the use of machinery on the farm but may contribute to reduce greenhouse gas outside the farm.

It should be considered to use the method to evaluate the impact of agricultural operations on more than a year in order to cover an entire rotation. The sequence of crops in the rotation has an impact on the operations and ultimately on greenhouse gas emissions and other impact categories. For example, analyzing operations for an entire rotation could solve interrogations like, in what year and on what culture is associated the environmental impact of seeding a cover crop?

5. Conclusion

This paper presented the approach used to develop a model predicting fuel consumption according to the different agricultural practices and the results obtained for wheat production.

Results showed that the largest variations were caused, not by production systems, but by soil preparation and fertilization operations. Average fuel consumption was 55.6 L/ha regardless of the production system, the soil preparation and fertilization strategies and the type of wheat. On average, soil preparation consumes 32% of the fuel, but it can reach more than 50% if conventional tillage is used. Grouped by tillage system, instead of production systems, no-till plots consume 39.6 L/ha, reduce tillage 55.6 L/ha and conventional 76.0 L/ha. Fuel consumption due to fertilization varies according to the type of fertilizer used. The amount of fuel used seems to be related to the concentration of nitrogen in the fertilizer. Winter wheat production used less fuel than spring wheat. Differences were observed because more producers used no-till and fertilized with manure.

6. References

ASABE (2011) Agricultural Machinery Management Data. ASAE D497.7 MAR2011

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Soil-carbon in LCA: Feasibility of measurement and monitoring systems

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ABSTRACT

In current LCA guidelines and most LCA studies of land-based systems, soil carbon is the "elephant in the room". Taking it into account is essential for many reasons, and yet it is still ignored. The lack of soil carbon data, as well as the site specificity of soil carbon stocks, makes the estimates highly uncertain. Actual monitoring of soil-carbon stocks is therefore necessary. In this study, we show that monitoring soil carbon at the scale of an apple orchard block is practically feasible at a cost of less than 1% of the price of export apples at the ship side. Furthermore, LCA studies should use measured, site specific, soil-carbon stock input data if meaningful results are sought. For transparency, the related uncertainty should always be expressed. Finally several actions for all stakeholders are presented for the implementation of soil-carbon stock monitoring.

Keywords: LCA, soil carbon, carbon footprint, monitoring, land use

1. Introduction

Soil-carbon is essential to maintain and enhance soil fertility; it also makes a contribution to many other ecosystem services (Victoria et al. 2012). Furthermore, the temporal changes in soil-carbon stocks could play an important role in mitigating climate change (Lal 2004). Therefore, soil-carbon change has been suggested as the main indicator in Land Use Impact Assessment in LCA (Koellner et al. 2013). However, most LCA studies include an incomplete consideration of the carbon balance because they omit total soil-carbon stocks and their changes in the calculations (Hamelin et al. 2012). Existing Life Cycle Assessments (LCA) and Carbon Footprint (CF) studies that account for or discuss soil-carbon stocks and their temporal changes suggest that the inclusion of soil-carbon can have a huge impact on the climate change results for land-based products. This effect can be over 100% for a change in land management (Land Use, LU) (e.g. Galdos et al. 2010) and over 2000% for land conversion (Land Use change, LUC) (e.g. Gelfand et al. 2013). Despite this substantial impact and its potential consequences for interpretation, LCA and CF guidelines such as the recent ISO TS 14067 generally exclude soil-carbon from the compulsory part of the assessment. As a result, many LCA and CF studies have excluded this essential parameter when assessing the impact of land based systems on the environment (Plassmann et al. 2010).

However, studies that consider or discuss soil-carbon change in the context of LCA and CF highlight three major issues. Firstly, there is a critical lack of soil-carbon data for all land-based systems and at all spatial scales (e.g. Smith et al. 2012). Secondly, they report the enormous uncertainty (e.g. Malca and Freire 2012; Meisterling et al. 2009), although rarely quantified, associated with the use of available soil-carbon data. Thirdly they demonstrate the site-specificity of soil-carbon stocks and their temporal changes, due to the complexity and inherent variability within and between land based systems (e.g. Bessou et al. 2011).

The role of soil-carbon stocks and their temporal changes in land-based systems as well as their potential influence on LCA and CF results suggest that soil-carbon should always be taken into account in LCA and CF studies assessing land-based systems. Furthermore, the important issues highlighted in the literature show the necessity of collecting site-specific data on soil-carbon stocks and changes and monitoring these changes over time.

The main objectives of this research were:

- To determine the practical and economic feasibility of monitoring changes in soil-carbon stocks at the field scale in commercial apple orchards.
- To propose guidelines for integrating soil-carbon in LCA, as well as actions for researchers, LCA specialists, businesses and policy makers to enable soil-carbon stock monitoring in agriculture, in order to improve the reliability and credibility of soil-carbon accounting in LCA.

2. Methods

Four commercial apple orchard blocks of different ages (Latitude: -39.588125, Longitude: 176.800076), were intensively sampled for soil-carbon to one meter depth to determine the spatial variability, and to assess the sampling requirements to observe a statistically significant change over time. The four apple orchard blocks of a chrono-sequence were chosen to be as similar as possible, in an effort to minimize non-age differences between these blocks. The apple variety, as well as the rootstocks, the soil texture to one meter depth and the depth of the water table were similar for the four orchard blocks. Furthermore, the management of the orchard blocks was very similar, being integrated fruit production, similar irrigation, and similar treatment at replanting. Finally, the previous land use was similar for the four orchard blocks. All blocks had been converted from grazing pasture and planted with apple trees more than 20 years before sampling, and more than 30 years for the four-year-old block. Figure 1 shows the spatial arrangement of the orchard blocks, to emphasize their proximity.

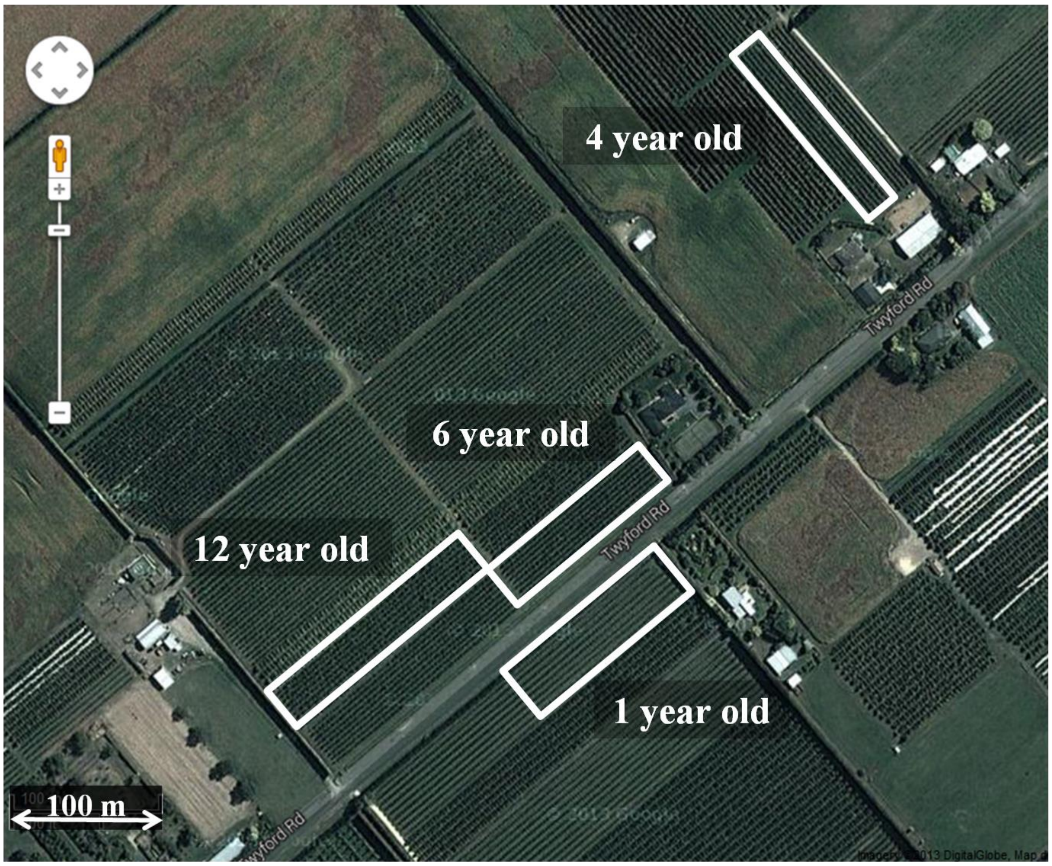

Figure 1: Map showing the proximity of the four orchard blocks. All blocks are within 500m of each other.

2.1. Sampling procedure

Within each orchard block, 10 sampling sites, or clusters, were randomly selected. For each sampling cluster, one bulk density (BD) profile (10 cm depth increments to one meter depth) was sampled. Furthermore, 16 soil samples to one meter depth were laid out in a 4 by 4 systematic grid within each sampling site. All %C samples were collected between 16/04/2012 and 24/08/2012. All BD samples were collected as soon as practicably possible, and within approximately one year of the % C samples. Following collection, all samples were kept in a cold room below 4 °C until further processed. The %C samples were sieved with a 2mm mesh and the fine-earth fraction was sent to a laboratory for total carbon content determination using a LECO TruSpec CN analyser (LECO Corporation, St. Joseph, Michigan, USA). The BD samples were oven dried at 105 °C until constant weight. Stones, roots and pumice rocks were removed and weighed separately for soil bulk density correction, in order to determine the weight of soil per unit volume.

The total carbon stock of the fine earth fraction, $C\ stock$, in tons of carbon per hectare (tC/ha) was determined for each depth increment according to the following formula:

$$C\ stock = BD_{fine\ earth} \times \%C_{fine\ earth} \times depth \qquad \text{Eq. 1}$$

The Genstat software was for the statistical analyses to determine the statistical significance of differences between orchard blocks.

In order to calculate the minimum number of samples, n, to observe a given change, δ, in soil-carbon stocks , and assuming that the same variance is found when re-sampling, the following formula from Steel et al. (1997) was used:

$$n = \frac{(Z\alpha_{/2} + Z_\beta)^2 \sigma_D^2}{\delta^2} \qquad \text{Eq. 2}$$

Here $Z\alpha_{/2}$ and Z_β are the values of the Z table with α the type-I error (statistical significance, α=5%) and β the type-II error (statistical power=1- β=0.8), respectively and where σ_D^2 is the variance of the difference between the two sample sets.

Therefore, the calculated number of samples required to observe a statistically significant change in soil-carbon stocks of 10% over 20 years in the top meter of soil was used to determine the economic feasibility of monitoring changes in soil-carbon stocks at the field scale in commercial apple orchards. The cost of routine minimum sampling (2 sampling campaigns: at year 0 and year 20 and each costing NZ$1590) was balanced against the price of export New Zealand apples at the ship side, taken as NZ$20/ carton equivalent (18kg of apples). The cost of routine minimum sampling was also used to calculate the price that a ton of carbon on the carbon market should reach in order to compensate for the cost of the sampling campaign. This analysis guided the identification of actions for the implementation of soil-carbon stock monitoring, including guidelines of how to take soil-carbon into account in LCA.

3. Results & discussion

3.1. Soil-carbon stocks

As shown in Figure 2, the carbon stock of the top meter of soil did not follow a particular trend along the chrono-sequence i.e. with regard to the time since the establishment of the current apple trees. There was no statistically significant difference at the 5 % level between the 1-year-old, the 6-years-old and the 12-years-old orchard block. This can be interpreted in several ways. First, one can deduce that the differences between soil-carbon stocks over 12 years are too small and the rate of change is too slow compared to the sampling campaign carried out. In other words, not enough samples were taken in order to reach a high enough precision of the average soil-carbon stock for each of the orchard blocks that would allow detecting a statistically significant difference between them. Secondly, it is possible that the four orchard blocks are different in a way not captured by the experiment, even though many factors were thoroughly evaluated to ensure a high degree of similarity between the four orchard blocks. Lastly, it is possible that no change in soil-carbon stocks took place over the twelve years along the chronosequence. In this case, current management practices in these three orchard blocks achieve the maintenance of the soil-carbon stocks over time.

However the soil-carbon stock of the 4-year-old orchard block had a much higher soil-carbon stock than the three other orchard blocks and was therefore significantly different (in a statistical sense) from the other blocks at the 5 % level. This large difference suggests that something happened to this orchard block that is different from the three other blocks. Two phenomena are potentially responsible for this difference. First, the 4 year old orchard block was converted from dairy farming to an orchard at least 10 years earlier than the three other blocks. Due to the fact that root turnover in the soil represents the main input of carbon into the soil, especially at depth (Schmidt et al. 2011), it is possible that having trees on the land for a longer time period favored soil-carbon sequestration. Secondly, a flood occurred in the area in the mid 1980s, and the deposition of alluvial sediments that were probably rich in carbon could have modified the carbon content of the soil, therefore increasing the soil-carbon stock. However, this second possibility remains uncertain, as the apple grower is not sure if the flood reached this part of the orchard block. And so far, no aerial photograph of the area affected by the flood was found.

Finally, the variation in soil-carbon stocks within each orchard block is equivalent to two standard deviations and is represented by the error bars on Fig. 2. It is important to note that this variation can be very different for different orchard blocks, even though the same sampling procedure was carried out on all orchard blocks. This indicates that generalizing a particular level of variation to any orchard blocks, from the four orchard blocks in this study, is not possible. In other words, each orchard block is affected by its specific level of variation, and sampling at this level is necessary in order to estimate it.

Overall, very different soil-carbon stocks with very different levels of variability were found in very similar orchard blocks, situated within less than 500m of each other. This has important implications for soil-carbon monitoring: The site specific factors, including the long-term land-use history for management and natural events means that monitoring soil-carbon stocks should be done specifically at the orchard-block level. Furthermore, using an average for all the blocks, even within one orchard comprising several blocks of various characteristics (variety, rootstock, time since first planting), is most likely to show a variability too high to allow observing statistically significant differences, even over long time periods such as several decades.

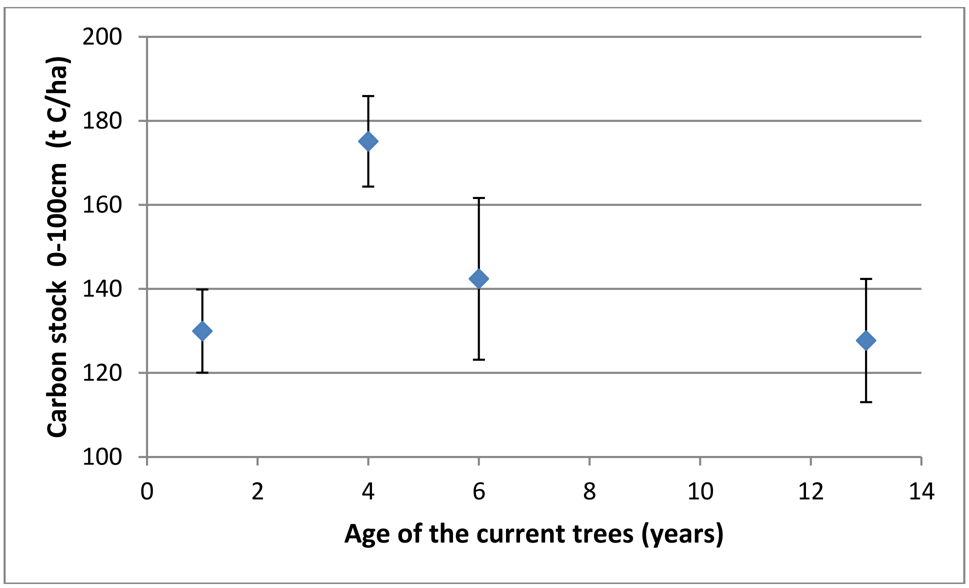

Figure 2: Chronological sequence of the soil-carbon stocks averages of the four orchard block. The error bars indicate two standard deviations (i.e. approx. 95% confidence interval)

3.2. Number of samples required

In order to estimate an indicative minimum sampling effort for monitoring, the 4 year old orchard block was selected for this calculation, as the variability of soil-carbon stocks within this orchard block is the smallest when compared relatively to its carbon stock. In other words, this orchard block has the smallest coefficient of variation in this study. The average soil-carbon stock for this orchard block is 175 tons of carbon per hectare.

In order to detect a 10% change over the top meter of soil, assuming an average rate of soil-carbon stock change of 0.5% per year, only 32 samples would be necessary and arranged in nine clusters with four samples per cluster. [This assumes that when sampling is repeated at the end of the 20 year period, the same level of variability compared to the initial sampling is found within the orchard block, which is uncertain, since root growth over this time period may affect soil-carbon distribution patterns.] However, observing a change of 5 % in the soil-carbon stock under the same conditions would require 240 samples (30 sites, eight samples per sites) for each sampling campaign. This highlights that monitoring soil-carbon stocks can be costly over short (less than 20 year) time periods.

3.3. Feasibility of monitoring

Several assumptions are necessary to determine the cost of sampling. First, it is assumed that a cost of NZ$1590 per sampling campaign can allow observing a change of 10% of the initial soil-carbon stock, and that the soil-carbon stock is measured twice, initially and 20 years later in the same orchard block. Secondly, it is assumed that an orchard block of one hectare that fits the first assumption, produces 50 tons of export quality apples per year from year 4 of the 20 years period. Thirdly, it is assumed that the sampling campaigns are conducted by a trusted independent organization, and no further sampling to verify the results is required. Fourthly, the value of apples at the ship side in New Zealand before export is assumed to be NZ$20 per carton (equivalent to 18kg) of apples.

According to these assumptions, which are believed by the authors to represent roughly the reality, one can calculate the value of a ton of carbon on the carbon market needed to compensate for the cost of monitoring, as well as the cost of monitoring associated with each carton of apples.

Value of a ton of carbon on the carbon market needed to compensate for the cost of monitoring:

$$= \frac{\text{Cost of two sampling campaigns}}{\text{tons of carbon stock change in the soil over 20 years (10\% of 175 tons)}}$$

$$= \frac{1590 \times 2}{17.5} = 182\text{NZ\$/ ton of carbon on the carbon market}$$

Cost of soil carbon monitoring per carton of apples:

$$= \frac{\text{Cost of two sampling campaigns}}{\text{number of cartons of export apples over the orchard block producing life (17 years)}}$$

$$= \frac{1590 \times 2}{\frac{50.10^3}{18} \times 17} = 0.07 \text{ NZ\$ per carton of apples}$$

This simple analysis shows that in order to compensate for the cost of monitoring soil-carbon stocks at the site specific level of an orchard block, relying on the carbon market is unrealistic, as it would require the price of carbon to be at least NZ$182 per ton of carbon. Even the optimistic forecast of the Intergovernmental Panel on Climate Change (IPCC) is based on a value of US$100/ton of carbon, which is roughly equivalent to NZ$120.

However, compensating for the cost of monitoring soil-carbon stocks every 20 years is largely feasible, as the price increase per carton of apples would have to be NZ$0.07, which represents less than 1% of the price of a carton of apples at the ship side. However, immediately this raises the question as to who should bear this extra cost. The benefits of monitoring soil-carbon reach far beyond the orchard, as soil carbon sequestration represents carbon taken out of the atmosphere, as well as improved soil health and fertility which helps keeping land use sustainable. It seems unfair that this cost should fall solely on the apple grower.

4. Conclusions

4.1. Implications for LCA

These results have three major implications for carbon footprint and LCA studies. First, perennial systems such as apple orchards may not always be associated with increases in soil carbon, as nevertheless suggested, for example, by reference documents such as the 2006 IPCC Guidelines for National Greenhouse Gas Inventories[1].

[1] Under Tier one of the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, the combination of soil-carbon stocks change factors relevant to the orchard blocks in this study (perennial cropland management, no till, medium residue input) indicate a carbon sequestration over time.

This study has indicated that no statistically significant change in soil carbon stocks could be observed over twelve years in a chronosequence of three orchard blocks. In addition, a decrease in soil-carbon stocks in perennial orchard systems is unlikely, due to the growth and turnover of tree roots, which represents the main input of carbon into the soil, especially at depth (Schmidt et al. 2011). As a general guideline, then, if site specific data on changes in soil-carbon stocks are not readily available, zero soil-carbon change should be assumed. Secondly, the fact that one orchard block of the chronosequence shows significantly different soil-carbon stocks compared to very similar and proximal orchard blocks demonstrates the high local specificity of soil-carbon stocks, and the high variability that can be presumed to underlie soil-carbon averages for land-based systems that are found in the literature. This suggests that ranges of data should be used to represent soil-carbon stocks and their changes and/or maintenance in LCA and carbon footprint studies. Thirdly, land-based systems that maintain soil-carbon stocks keep carbon out of the atmosphere. It is therefore reasonable to consider that, as well as accounting for changes in soil-carbon stocks, also the maintenance of soil-carbon stocks over time in agricultural systems should be recognized in LCA and carbon footprint studies.

4.2. Feasibility of soil carbon monitoring

Soil-carbon stocks monitoring can be realistically implemented, in the case of New Zealand apples, for less than 1% addition to the price of export apples at the ship side. The industry body seems to be the appropriate stakeholder to coordinate field monitoring and to communicate results to export retailers, because of their close interactions with both farmers and retailers. The implementation of a price premium for apples coming from an orchard block where soil carbon is monitored could be a solution to finance soil carbon monitoring. The cost would not be borne by apple growers, and would not involve politically unpopular measures such as making soil carbon monitoring a requirement for market access.

4.3. Actions for stakeholders to implement soil-carbon monitoring

The following actions would support implementation of soil-carbon monitoring:

- Researchers improve and develop standard, crop-specific, statistically representative, rapid and inexpensive measurement methods to collect data in order to capture the changes in soil-carbon stocks and parameterize biophysical models.
- LCA specialists favor and advocate using site specific, statistically representative soil carbon input data, as well as quantify and report the uncertainty associated with using non site-specific soil-carbon stock data from databases and literature in LCA studies, together with their potential impact on the LCA results.
- Industry bodies introduce and organize monitoring schemes as well as their promotion to export markets.
- Farmers are recognized for their efforts to monitor and maintain soil-carbon stocks.
- Policy makers and retailers facilitate and promote schemes such as price premiums for food products from land based systems involving monitoring, maintaining and improving soil-carbon stocks.

4.4. Guidelines to integrate soil-carbon in LCA

The maintenance of soil-carbon stocks needs to be considered as having a beneficial, rather than neutral, impact on the climate change and soil quality impact categories. Furthermore, using measured, site specific (and geo-localized) soil-carbon stock data in LCA is necessary if meaningful results are sought. Even nearly identical and proximal sites can have very different soil-carbon stocks as a result of different past land use and/or natural events. If this is not possible, and because of the large potential impacts of soil-carbon input data on LCA and CF results, the inherent variability and the uncertainty associated with literature or model estimates of soil-carbon stocks and changes should be quantified and communicated. Finally, these soil-carbon input data should be accompanied by sets of data describing the environmental and management conditions in which they were collected, or, by default, the related assumptions associated with the choice of a particular estimate. This will improve the transparency and the reliability of LCA results of land-based systems.

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Improving the accounting of land-based emissions in Carbon Footprint of agricultural products: comparison between IPCC Tier 1, Tier 2 and Tier 3 approaches

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ABSTRACT

In this paper, we discuss different methods to calculate greenhouse gas field emissions from fertilization and soil carbon changes to be integrated into Carbon Footprint (CFP) of food and biomass products. At regional level, the simple Tier 1 approach proposed in the IPCC (2006a) AFOLU guidelines is often insufficient to account for emission variability which depends on soil type, climate or crop management. However, the extensive data collection required by Tier 2 and 3 approaches is usually considered too complex and time consuming to be practicable in Life Cycle Assessment. We present four case studies to compare Tier 1 with medium-effort Tier 2 and 3 methodologies. Relevant differences were found: for annual crops, a higher Tier approach seems more appropriate to calculate fertilizer-induced field emissions, while for perennial crops the impact on CFP was negligible. To calculate emissions related to soil carbon change higher Tiers are always more appropriate.

Keywords: agricultural production, GHG accounting, regional variability, field emissions, soil carbon change

1. Introduction

During the last decade, the interest of companies and policy makers in Carbon Footprint (CFP) as a supporting tool to assess the global warming impact of food and biomass productive processes and to design impact reduction plans has grown. All greenhouse gas (GHG) emissions occurring within the established boundaries of such a study have to be taken into account, as instructed in the ISO 14067 (2013), the most recent international reference standard for the development of CFP studies.

Land-based CO_2 and N_2O emissions from fertilization and crop residues management can account for a considerable amount in the GHG balance of food and bioenergy products, together with CO_2 fluxes due to soil carbon change (Brentrup et al. 2000; Petersen et al. 2013), but they are often disregarded in CFP studies. IPCC guidelines for National GHG Inventories (IPCC 2006a) provide, within the fourth volume dedicated to Agriculture, Forestry and Land Use sector (AFOLU), three calculation pathways (Tiers) for the accounting of these land-based emissions characterized by different degrees of complexity: Tier 1 level includes the less accurate methodologies, which can be applied by using the provided global emission factors; the Tier 2 level methodologies require using emission factors specific for the region which is subject of the study, while Tier 3 level methodologies are based on measurements or simulations performed by models. The mentioned IPCC guidelines suggests always using a more accurate methodology if possible, and provides decision trees to support the identification of the suitable Tier.

At regional and sub-regional level, Tier 1 level methods are not always sufficiently accurate to account for geographical variation of emissions dependent on different soil, climate or management practices. Conversely, field measurements and extensive data collection required by higher Tiers (Tier 2 and Tier 3) are usually considered too complex and time consuming to be practicable in the development of Life Cycle Assessment (LCA) studies.

The first aim of this paper is to test assessment methods at Tier 2 and Tier 3 level to calculate land-based GHG emissions from crop cultivation with medium efforts for stakeholders. The selected Tier 2 method consists of calculating N_2O emissions from fertilization in consideration of pedoclimatic and crop management conditions based on Bouwman et al. (2002) and the Tier 3 method of simulating CO_2 fluxes occurring as consequence of soil carbon stock change based on Petersen et al. (2013). The second aim is to assess and compare the influence of the variability of regional inventory data on CFP results, depending on the adoption of Tier 1 or Tier 2 and Tier 3 accounting methods. We performed analysis in four case studies: two wheat croplands in Germany and one peach orchard in Italy managed with two different regimes. The choice of case studies was based on different soil characteristics, climate conditions and crop types (annual and perennial), to test how different methodologies may represent this variability.

2. Materials and Methods

2.1. Methodologies for the assessment of fertilization induced emissions on field

The simple IPCC Tier 1 method (IPCC 2006a) for calculating the emission of nitrous oxide (indirect and direct N_2O) from managed soils only takes into account 1% of the anthropogenic N inputs on the field. This approach disregards differences of crop type, fertilizer type, management system and local climate conditions. In contrast, considering all these agricultural relationships to calculate N_2O, NO and NH_3 emissions, the heterogeneity of environmental and management conditions occurring in agriculture would be reflected more accurately.

We have chosen the model approach from Bouwman et al. (2002) for direct and indirect N_2O emissions and the approach from FAO and IFA (2001) for NH_3 volatilization. These methods are more performant on the local scale and under different agricultural management systems to reduce the uncertainty range within the global emission Tier 1 factors (IPCC 2006a). Implementing this Tier 2 approach more detailed data are required. The multivariate empirical model of Bouwman et al. (2002) divides the parameters which have an influence on the N_2O and NO emissions into specific categories (fertilizer type and application rate, crop type, soil texture, soil organic carbon (SOC), soil drainage, soil pH and climate type), for each factor. NO and NH_3 emissions were converted to N_2O based on IPCC (2006a) by the factor 0.01. For the NH_3 emissions induced from organic fertilizer (mature and liquid manure, digestate, poultry manure) we made an exception and used the model approach based on KTBL (2009). CO_2 emissions from the application of urea and liming on soil were calculated based on Tier 1 IPCC (2006a) factors.

2.2. Methodologies for the assessment of emissions from soil carbon stock change

To include the soil carbon change into the CFP accounting the Tier 1 and the Tier 3 methodology have been tested and compared.

The simple Tier 1 is explained in Chapter 5 of Volume 4 (cropland remaining cropland) of IPCC guidelines (IPCC 2006a). The soil organic carbon reference (SOCref) under native vegetation, has to be selected depending on six soil types and nine climate regions, together with three relative stock change factors: F_{LU} related to land use, F_{MG} related to tillage regime and F_I related to carbon input level. These factors are characterized by different error ranges (between ±5% and ±50%) and have to be selected, based on the climate region, for both conditions before and after the management or land use change occur.

The Tier 3 methodology consists in a simulation of the turnover of carbon in soil with the model Roth C 26.3 (Coleman and Jenkinson 1999), integrated with the Bern Cycle Model as suggested by Petersen et al. (2013); the Bern Cycle Model (IPCC 2007) simulates the decay of CO_2 in the atmosphere in order to represent the release of CO_2 into the atmosphere over several years. The outputs of the Roth C simulation are the CO_2 emissions and removals related to soil organic carbon change. The Roth C simulations were run for both scenarios (baseline and changed) for a number of years sufficient to reach the equilibrium (n), and the resulted emissions and removals were divided by the same number of years in order to obtain the yearly figures. The following equation was used to calculate the Tier 3 results:

$$\Delta CO_{2\,(n)}\ [t\ CO_2\ year^{-1}] = [(CO_2\ E_{CS} - CO_2\ E_{BS}) + (CO_2\ R_{CS} - CO_2\ R_{BS})]/n \qquad \text{Eq.1}$$

where $\Delta CO_{2\,(n)}$ represents the net balance of CO_2 in the atmosphere after *n* years, related to crop management change; E_{CS} and E_{BS} are the emissions resulting from soil organic matter decomposition after *n* years, respectively in changed scenario and in baseline scenario; R_{CS} and R_{BS} are the removals resulting from carbon stored in soil after *n* years in the changed scenario and in the baseline scenario.

2.3. Field experiments

Four field trials were selected based on crop type (annual crops and perennial crops), soil and climate conditions. Characteristics of experimental sites are presented in Table 1. The two winter wheat experiments were cultivated as sole-food-crop at two different sites in Germany (Site 1 and 2). They were sown at the end of September after ploughing and harvested in the following summer (end of June), leaving the straw at field. The rate of nitrogen fertilizer was determined site-specifically and split into 50% mineral and 50% organic N-fertilizer (digestate).

The perennial crop field trials were conducted on a peach orchard located in Southern Italy (Site 3). The orchard was divided in two plots, one cultivated with local management practices and one converted to improved management practices. The local management regime consisted of: no tillage, micro-jet sprinkler irrigation, pruning residues mulched and left on field, chemical weed control with glyphosate and mineral fertilization based on experience. The improved management regime introduced some innovative: drip irrigation based on daily water balance, spontaneous grass cover mowed twice per year, precision fertilization based on periodic monitoring of soil nutrient availability and organic fertilization with 10 tons of compost per hectare and year.

We assumed that no land use change occurred in our four case studies, but the cultivation management changed. In the winter wheat case studies the straw management was changed. Before the change all straw was harvested to be used for energy production or animal feeding. However, after the change it was left at field and incorporated later. In peach orchard experimental site managed with local regime the change occurred after the orchard establishment, when pruning residues started to be incorporated in the field instead of being burned (as in the past orchard). In the plot converted to improved management one more change occurred: compost and grass cover were added to the soil to increase the quantity of the carbon input comparing to the local management regime.

Table 1: Characteristics of experimental sites

Site number	**1**	**2**	**3 (Local and Improved)**
Crop	winter wheat (*Triticum aestivum L.*)	winter wheat (*Triticum aestivum L.*)	Peach (*Prunus Paersica*) Variety Big Bang/ GF/677 Trained to delayed vase Layout: 5,50 x 4
Name	Ascha	Dornburg	Scanzano Jonico
Period of data collection	2011-2013	2011-2013	2006-2013
Country	Germany (south)	Germany (central)	Italy (south)
Geographical location	48°59´N 12°39´E	51°00´N 11°39´E	40°14´N 16°42´E
Height above sea level (m)	431	247	16
Soil type	Stagnic Cambisol	Luvisol	Eutric Vertisols
Soil texture	Loamy sand	Silty clayey loam	Sandy clay loam
pH-value	5,1	7,4	7,4
Bulk density (g/cm³)	1,7	1,5	1,5
SOC (%)	0,5	0,5	1,3
Total annual precipitation (mm) (long term mean)	957	582	550
Mean temperature (°C)	8,3	9,6	16,8

2.4. Goal and scope, Life Cycle Inventory and Impact Assessment

Greenhouse gas emissions of the four crop cultivations were investigated according to the standards ISO 14040 (2006), ISO 14044 (2006) and ISO 14067 (2013).

The focus in the case studies was on the agricultural activities during the food production, which is why the selected functional unit was the unit of cropland (1 hectare). System boundaries were fixed from cradle to farm gate, starting with production of all inputs (seeds, fertilizers, pesticides, agricultural machinery and fuels) and ending with harvesting the crop, encompassing all direct and indirect emissions

For peach case studies, the whole life cycle of the orchard has been included within the time boundaries, from the first year of orchard establishment till the last year before removal, coherently with the most common practice of LCA sectorial studies about fruit production from perennial tree crops (Cerutti et al. 2010; Milá i

Canals and Polo 2003). The considered impact category was the Global Warming Potential (GWP) – 100 years with the characterization factors from the "CML, 2001" method (Ecoinvent 2013).

The calculation of CFP have been divided in three main parts: land based emissions from fertilization, CO_2 fluxes from soil carbon change, and all other impacts from agricultural operations.

Consequences of methodological choices have been analyzed by comparing the CFP results based on IPCC Tier 2 and Tier 3 approach with the reference CFP results based on IPCC Tier 1.

3. Results and discussion

Applying the approach from Petersen et al. (2013) it is possible to evaluate if the change in crop management practices results or not in a net saving in terms of CO_2 released to the atmosphere. For the wheat case studies (Site 1 and 2), the CO_2 removed from atmosphere caused by soil carbon stock change is much lower using Tier 3 than using Tier 1. This means that for changes in management practices of annual crops (straw added to soil) the Tier 1 overestimates the benefits in terms of carbon stored in soil (Figure 1 B). This result is determined by the stock change factor related to carbon input level F_I. Performing the calculation with Tier 3, the real quantity of straw added to soil is considered, which results in a lower saving of CO_2 linked to soil carbon stock change. In both case studies of winter wheat (site 1 and 2) the Tier 3 estimates are outside the error range indicated in Tier 1.

For the peach orchard case study the situation is opposite. The amount of CO_2 stored in soil is much higher using Tier 3 than using Tier 1(Figure 1 D). The stock change factors from Tier 1 are not sufficient to represent the real entity of soil carbon change in a perennial crop. Even if the Roth C model does not consider the effect of tillage on CO_2 emission from soil organic matter decomposition, the Tier 3 methodology gives higher results in terms of soil carbon change than Tier 1 in the case of perennial crops. In the peach case studies the results of Tier 3 methodology are inside the error range forecasted for Tier 1.

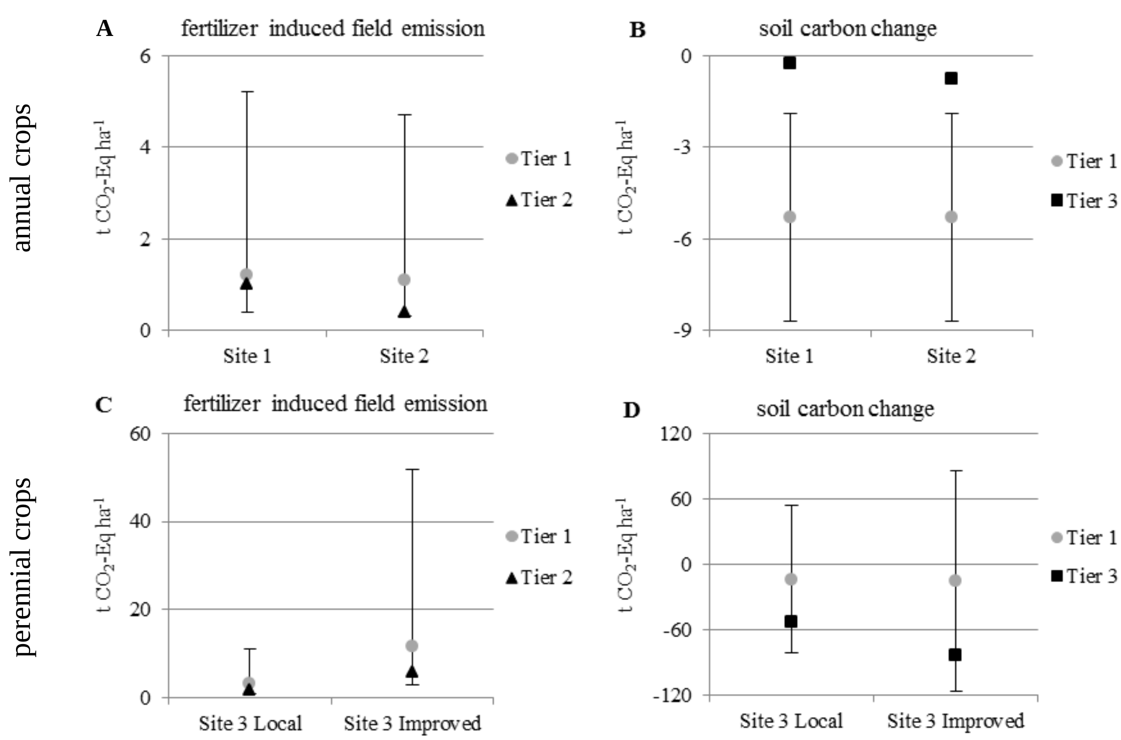

Figure 1. Comparison of the emissions from fertilizer applied on field and soil carbon change calculated with Tier 1 and Tier 2/3 approaches for the four case studies (Site 1 and 2: mean values from two cultivation years at each site, Site 3: values are summed up over 13 years)

For annual crops (wheat cropland), field emission from fertilizer application is an important factor, accounting for almost 50% of the whole CFP-emissions calculated with Tier 1 at Site 1 and 2. In contrast, for perennial crops (peach orchard) the amount of emissions from fertilizer application is less than 25%. Using a higher Tier (Bouwman et al. 2002) to calculate GHG emissions from fertilizer application on field, the CFP is reduced by −8% for wheat production in Site 1, −28% for wheat production in Site 2, −6% for peach production in Site 3 with local management and −12% with improved management. Looking at the Tier 2 fertilizer induced field emissions (Figure 1 A, C), the estimated values are in any case study lower than the Tier 1 approach, and are within the uncertainty range of ~ -70% to +350% reported for the default global emission factor from Tier 1.

As reported in the IPCC (2006a) guidelines, in many cases the global default values (Tier 1) are adequate to determine the field emissions like in one of our wheat case study (Site 1). But in most cases these factors have to be more specified based on environmental conditions (climate and soil characteristics) like in our wheat case (Site 2) as well as on crop management conditions like in our peach cases (Site 3 local and improved management). Considering these fertilizers induced field emissions a higher Tier approach will allow to detect more specific mitigation potentials.

4. Conclusion

We identified appropriate assessment methods on Tier 2 and Tier 3 approach level with medium efforts for stakeholders and explored the consequences of these methodological choices on CFPs of annual and perennial crops for land-based GHG emissions from crop cultivation. The results for fertilizer induced field emission calculation were consistent among studies, using the higher Tier (Tier 2) led to a reduction of the GHG emissions. We have observed that using the Tier 1 approach overestimates the field emissions from fertilization.

To include field emissions related to soil carbon stock change into CFP of agricultural products is challenging. A wide difference was found between results calculated with Tier 1 and Tier 3 methodologies.

A more shared consensus within the LCA practitioners community is needed about how to perform the calculation using higher Tiers, in order to improve the assessment of agriculture´s mitigation potential and support the development of GHG reduction plans in the primary sector. Using the Tier 1 approach can lead to wrong estimations, due to the qualitative nature of the relative stock change factor. This is why we suggest using the Tier 3 approach for perennial crops where CO_2 emission savings related to soil carbon change are more relevant and for annual crops to avoid an overestimation caused by the use of Tier 1.

Variability in methods can considerably affect the CFP from agricultural productions. A harmonization and more transparency in methods are necessary to distinguish between actual differences of case studies and differences caused by methods. For annual crops the higher Tier approach is very important to perform the fertilizer induced field emissions, for perennial crops on the other side the fertilizer induced field emissions have just a minor impact on the CFP. The soil carbon change is negligible because of the short cultivation period of annual crops. However, for perennial crops the soil carbon change should be considered in the CFP. Only a few more site-specific data are needed to perform these higher Tier approaches, which can be used in both methodological improvements.

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Improvement of odor assessment in a life cycle assessment framework

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ABSTRACT

Odorous emissions are a key concern for intensive livestock industries, wastewater treatment and other industries due to growth and increasing overlap with urban areas. Relatively little effort has gone into the development of methods for odor within the field of life cycle assessment (LCA). Traditionally, the mass of gas emitted and its detection limit by humans are considered. So there is a simple effect model, but no transport model. Detailed approaches for odor transport modeling exist outside LCA but they are difficult to apply in strategic environmental assessments. To support debate around the development of a midpoint indicator with transport aspects for odor, we calculated characterization factors for an "odor footprint", using data on the rates of diffusion and reaction to model persistence and spreading. We suggest that a simplified approach to transport modeling is reasonable within the LCA context.

Keywords: odour, footprint, midpoint

1. Introduction

Many food production systems involve components that generate odor, particularly those where animal wastes are involved. Management systems for some textile factory wastewaters, sewage, sludge and other organic materials have similar challenges exacerbated by urban encroachment. Odor entering domestic and commercial buildings reduces the amenity of indoor working and home environments. Strategic environmental comparisons between production systems or between odor management systems should preferably take odor into account but approaches for doing this in a life cycle assessment (LCA) framework have received little developmental attention over the last 20 years. Perhaps the first attempt was described in the original "Guide and Backgrounds" to LCA (Heijungs, 1992) where odor threshold values (OTVs) were the key element in a life cycle impact assessment (LCIA) for "malodorous air", a characterization approach based on critical volumes. The OTV is the concentration ($kg.m^{-3}$) at which a chemical is detectable by 50% of the population. Thus, for the mass *m* of each odorant *i*:

$$\text{malodorous air} = \sum_i \frac{m_{i,air}}{OTV_{i,air}}$$

What this means is that an emission of a mass of an odorant is characterized as the volume of air it would occupy if instantly diluted to its OTV. Heijungs (1992) proposed that this could be the basis for future development of a "smell creation potential" indicator, including transport and fate considerations. But this has been a long time coming: Heijungs' original approach and the 60 characterization factors (CFs) he obtained from an older document (Roos, 1989) were duplicated in the subsequent Nordic LCA guidelines (Nord, 1995) and the updated Backgrounds document (Guinee, 2002). The method is suggested (without providing data) in the popular Hitch Hiker's Guide to LCA (Baumann and Tillman, 2004). The recent International Life Cycle Data System (ILCD) Handbook (JRC, 2010), does not provide any guidance on odor, so the most recent consensus guidance documents on odor use a life cycle impact assessment (LCIA) approach that has not changed in 22 years, using data which is 25 years old. One could call Heijungs' method the established approach since it has been reproduced as described, and has been used in several LCAs, but with the qualification that no odor method has been widely applied in the LCA literature.

Perhaps the most interesting thing to happen recently in this area is a proposal to modify the USEtox model as a basis for fate modeling of odorants (Marchand et al., 2013). USEtox is a consensus-based box-model for the calculation of LCIA factors for chemical contaminants, but it is feasible to extract steady-state concentrations for different atmospheric compartments from it and compare these with OTVs. Marchand's proposal involves a number of adjustments to the environmental descriptors in the standard USEtox model to make it more relevant to the environmental assessment of a particular local area. It also suggests OTVs should be placed on a fuzzy scale, with potential effects below the nominal OTV. Another innovation on Heijung's approach was the suggestion that the ultimate odor burden be expressed in terms of 11 different midpoint indicators, one for each different type of smell (sweet, rancid, fecal etc.).

Balancing parsimony with accuracy is an important consideration in the development of LCIA (Hauschild et al., 2008). It is a consideration which is the more challenging when LCIA attempts to characterize emissions with local (as distinct from global) impacts. To encourage further discussion around this balance, the aim of this work was to develop a simple method for odor midpoint assessment in LCA.

2. Methods

We considered the cause-effect sequence between emission of an odorant and the reduction of the value of the protection objects typically included in LCA: human health, ecological systems and resources (e.g.: Kounina et al., 2013). That overview suggested the potential for a midpoint of "odor footprint" which does not represent the actual area disturbed by odor, but the relative potential of different emissions to cause odor problems. A more detailed description of the method will be provided in a journal article recently accepted after peer review (Peters et al., 201x) but salient details are summarized here.

The key decision to be made in developing an odor LCIA method may be what approach to take to wind dispersion. In normal odor dispersion modeling as applied to the assessment of single and mixed odorants, wind speed is a key input in the calculations and also determines dispersion parameters. The time step in the input meteorological data and subsequent modeling may be hours or even shorter intervals. Such dispersion models are commonly used in regulatory and detailed design processes, for example when decision-makers need to assess whether a particular emission will affect a particular urban area. We felt a simpler approach could be justified in an LCA context, as illustrated in the following example. Imagine that a landowner is considering construction of a facility on her land and commissions an LCA comparing production system 'A' which emits odorant 'X' and system 'B' that emits 'Y'. In either case, the same wind will blow, so the diluting effect of the wind on 1 kg of X or Y will be the same. On the other hand, degradation and diffusion rates, and odorant potency (OTV) may distinguish the impacts of X and Y, as these are based on inherent chemical properties. Since LCAs are often performed to compare alternatives, making a *relative* odor impact scale relevant, we chose to take these three factors into account and exclude wind. This will not replace the need for absolute assessments based on dispersion modeling for environmental regulation or detailed design, but may fill the need for strategic LCA assessments.

The approach to odor modeling in this article is illustrated by Figure 1. An initial mass of 1 kg of odorant is assumed to be uniformly distributed within a 1 m diameter hemisphere at ground level (the dashed volume outline in the figure). The gas is allowed to diffuse radially (path 'a' along axis 'r' in the figure), thus the land area at which the concentration exceeds the OTV increases. The increase is counteracted by simultaneous degradation reaction with hydroxyl radicals. Although certain gases may react with other anthropogenic and natural gaseous reagents, the most important route for the destruction of organic odorants is reaction with hydroxyl radicals (Kwok and Atkinson, 1995). The area exceeding the OTV reaches a maximum radius (the solid outline) and then collapses back to the origin (path 'b') due to the continuing degradation process.

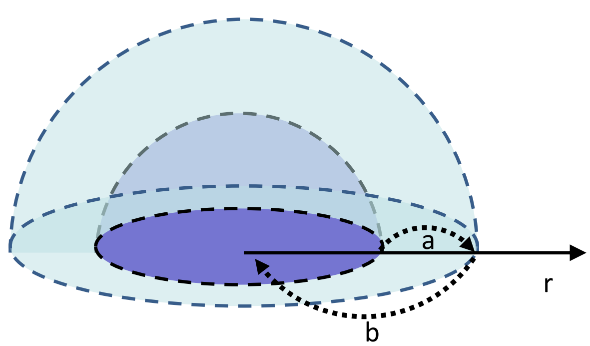

Figure 1. Odor modeling approach

For the calculations, OTV values were taken from Nagata (2003) who measured values for 223 gases using the triangle odor bag method. We obtained estimates of k_{OH} from the NIST database (Manion et al., 2008) and estimated D, the Brownian diffusion coefficient, using Fuller's method (Onken et al., 2008). Atmospheric pressure and 25°C were assumed, consistent with a typical daytime value of the atmospheric concentration of hydroxyl radicals [$^{\cdot}OH$] of 1.5 x 10^6 molecules.cm^{-3} (Allen, 2001). Computational modeling was implemented in Matlab® version 2012a.

3. Results

Figure 2 shows an example of the model output. Hydrogen sulfide diffuses approximately twice as fast as methyl ethyl ketone, and its OTV is a factor of 2000 lower. Both of these factors lead to the steeper, taller curve. The degradation rate of methyl ethyl ketone is a quarter of that of hydrogen sulfide, leading to the longer persistence of concentrations exceeding the OTV. The integral of the curve is the cumulative area x time the OTV is exceeded, i.e. the odor footprint.

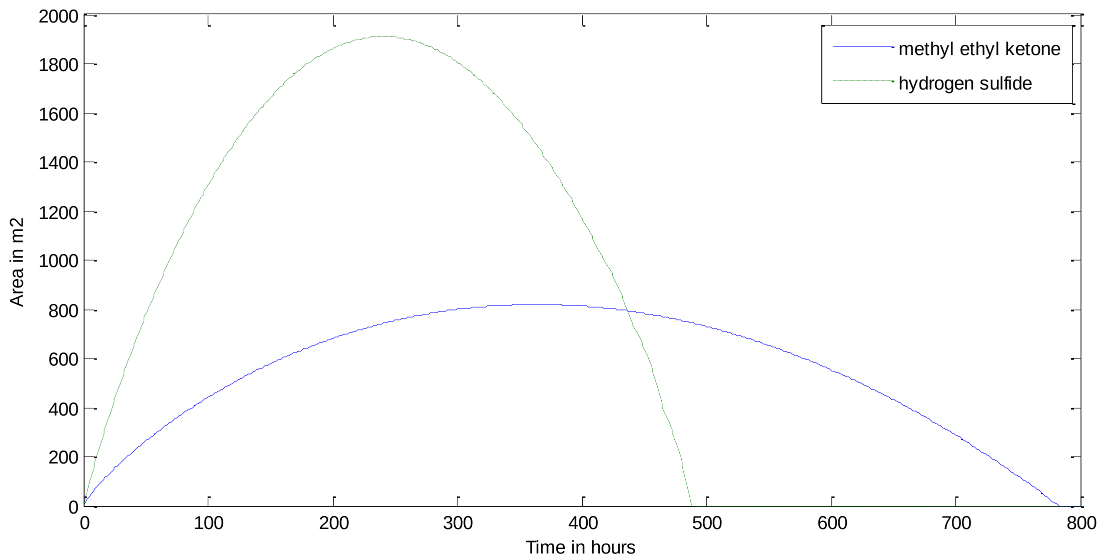

Figure 2. Illustration of footprint growth and decay over time

We have calculated characterization factors for 33 key odorants prioritized on the basis of their presence in LCA literature and the literature on emissions from agriculture and wastewater treatment (Peters et al., 201x). Two of these results are shown in Table 1. As can be seen from this table, the relative significance of a kilogram of the two contaminants is quite different if the analyst was to use Heijungs' method: a kilogram of hydrogen sulfide is equivalent to a volume of "malodorous air" 2267 times greater than a kilogram of butanone, while the odor footprint of the hydrogen sulfide is only 46% larger. The difference in these relationships is due to the more rapid decay of hydrogen sulfide which the new method considers.

Table 1. Sample results from characterization factor estimation

Parameter	**OTV**	**D**[1]	**k_{OH}**	**Odor footprint**	
Odorant	$g.m^{-3}$	$m^2.s$	s^{-1}	m^2s	kg H_2S-eq.kg^{-1}
Hydrogen sulfide	$5.72 x 10^{-7}$	$2.31 x 10^{-5}$	$7.05 x 10^{-6}$	$2.30 x 10^{9}$	1.00
Butanone (methyl ethyl ketone)	$1.30 x 10^{-3}$	$1.29 x 10^{-5}$	$1.80 x 10^{-6}$	$1.57 x 10^{9}$	$6.83 x 10^{-1}$

[1] Diffusion coefficient

4. Discussion

The expression “midpoint indicator” does not exactly describe the location of an indicator on the cause-effect chain between an emission and an impact on a protection object. Both the established metric for odor LCIA of Heijungs (1992) and the recent proposal of Marchand et al. (2013) could be called midpoint indicators, given that neither of them computes an output that is expressed in the terms of common endpoint indicators for human health (e.g.: DALYs), ecosystem quality (e.g.: PDF.m^{-2}.$year^{-1}$) or resources (e.g.: MJ). As suggested by Table 2, our proposal lies somewhere between these two in terms of the level of detail used in the modeling. An analogy can be made between our proposal and the characterization factors for the popular ”carbon footprint” indicator, which do not predict potentially nonlinear consequences of climate change like species extinctions and human fatalities due to sea-level rises, but rather a change in radiative forcing – the gases’ relative potential to cause these effects. This odor footprint model compares odorant’s relative potential to affect air quality, rather than the likelihood of a particular, geographically defined receptor being disturbed.

Table 2. Comparison between established and more recent proposals for odor LCIA

	Heijungs, 1992	Marchand et al., 2013	This proposal
Output	Single indicator: “malodorous air” in m^3	11 indicators (kg-equivalents) based on a key odorant for different smell types	Single indicator: “odor footprint” in kg H_2S equivalents
Transport model	Gas dilutes instantly to OTV	Modified USEtox approach including degradation and removal by a constant wind.	Considers degradation and diffusion
Effect model	Comparison with OTV	Fuzzy interval (uniform factor of 1000) as a safety margin for effects below OTV	Comparison with OTV
Practical operation	Elemental flows easily divided by characterization (OTV) factors	11 factors adjusted in USEtox spreadsheet for each emission location. Model run for each odorant at each location. Steady-state concentrations extracted individually for comparison with effect model.	Elemental flows easily multiplied by characterization (odor footprint) factors.

The new approach we propose has the advantage over Heijungs’ approach of considering the temporal aspect of odorant impact and the advantage over Marchand’s approach of simpler implementation. The question we would like to answer in future research is whether modification of USEtox to describe odorant dilution at a facility and local scale is more meaningful and/or accurate than our simpler approach. Marchand provides an example based on ethylbenzene emissions from a composting facility, which includes initial dilution in a tall rectangular airspace 50 m square at the base and 240 m high, before subsequent dilution in a 6 km square and 1 km high, the latter “based on the average dispersion area of odorous compounds”. In practice, the steady-state concentrations in the facility and local USEtox compartments are directly inversely proportional to volume. On the other hand, if one tests the sensitivity of USEtox by halving and doubling wind speed, rain rate or temperature (factors which Marchand suggests are the most sensitive) a much smaller effect on the steady-state concentration in the local air compartment is observed. This suggests that if adjustments are made to the scale of USEtox compartments in future studies which are not justified by first performing detailed odor dispersion modeling, then comparisons of odorous systems using the results generated by USEtox are unlikely to be more accurate than they would be using the proposed odor footprint method. Alternatively, if detailed odor dispersion modeling is performed for scaling purposes, there may be no need to assess odor in an LCA framework.

Another matter to consider when implementing any of the methods is the appropriate source of OTV data. The available OTV data has been augmented considerably since 1989. Furthermore, a transcription error occurred during production of the Nordic Guidelines (Nord, 1995) so the values in the bottom half of the list in the Guidelines are too large by a factor of 1000. So even if an analyst wishes to use Heijungs’ method, it would be more appropriate to use it with the Nagata (2003) values. For example, in comparison to the data shown in Table 1, the original values of the OTV for hydrogen sulfide and butanone are quite different, 4.3×10^{-7} and 6.8×10^{-4}

$g.m^{-3}$, respectively. The new compilation is superior compared to other available compilations of olfactometry results because the OTVs are corrected for recovery of single compounds and losses in the dynamic olfactometer are avoided (Hansen et al. 2013). Additionally, unlike other compilations of OTVs, the Nagata data all are obtained in the same lab using the same method. These factors are reasons why recent work on OTV prediction by Abraham et al. (2012) is also based on the data we obtained from Nagata (2003).

5. Conclusion

The established method for odor assessment in LCA effectively places emphasis on the detectability of an odor by the human nose. No consideration for the persistence of an odor impact is made. Our proposal is a way of including persistence in LCIA of odorants and gives more weight to slower reagents. We suggest that in practice, alternatives using USEtox to model odors for LCA purposes may lead either to unnecessary work or false precision, depending on whether or not the analyst has previously implemented site-specific dispersion modeling to assist in setting local parameters in USEtox.

The key difference between the LCA odor footprint proposed here and local odor impact prediction using dispersion models is that the disposition of sources and receptors is not known in LCA, so the influence of wind is excluded – the dispersion effect of wind dilutes all odorants equally. So while an odor footprint will not predict impacts which are particular to a site or climate (endpoint modeling), it is an improvement on the traditional LCA approach for ranking odors.

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An "LCA" approach to Slow Food Presidia products: from agro-environmental and socio-cultural aspects to economic sustainability and nutritional evaluations

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ABSTRACT

Two approaches, nutritional and multi-criteria sustainability, were combined to define Life Cycle Assessment (LCA) of Slow Food Presidia in a new way. Chemical and bromatological evaluation of several plant and animal Presidia products were compared to traditional Italian bromatological references. To assess sustainability, producers were interviewed on four objective areas: environmental, economic, social and cultural. The gathered data were used to value Presidia.

Keywords: Food LCA, Sustainability, Slow Food Presidia Nutritional Value

1. Introduction

Slow Food is a global, grassroots organization, founded in 1989 to prevent the disappearance of local food cultures and traditions. It is also intended to counteract the fast pace of life and people's dwindling interest in the foods they consume, including their sources, production, and how their choices affect the world around us.

Since its inception, Slow Food has grown into a global movement involving millions through its activities:

- saving endangered foods and defending gastronomic traditions through biodiversity projects;
- teaching food pleasure and how to make good, clean, and fair choices through food and taste education;
- connecting young people passionate about changing the food system via the Slow Food Youth network;
- organizing countless daily activities for local Slow Food member groups (convivial) on food, the environment, and food-related sustainability;
- linking food producers, chefs, academics, and community representatives across the world through the Terra Madre network;
- creating the next generation of food and gastronomy professionals at the University of Gastronomic Sciences.

The Slow Food Foundation for Biodiversity was founded in 2003 to support Slow Food projects, defending food biodiversity and traditions. The Presidia project started in 1999 with a recording of hundreds of products facing extinction through the Ark of Taste. Thereafter, Slow Food engaged the production side of the foods (the where, how, and who) to promote the products, work, and wisdom.

The Presidia sustain their production quality while risking extinction of their unique regions and ecosystems, traditional processing methods, native breeds and local plant varieties. There are more than 200 Presidia in Italy and more than 170 International Presidia, involving 2500 small-scale fishers, butchers, shepherds, cheese makers, bakers, and pastry chefs.

2. Methods

2.1. The project

Since 2009, the Slow Food Foundation for Biodiversity (SFFB) in collaboration with Turin and Pollenzo Universities and the Chemical Laboratory of the Chamber of Commerce of Turin, has started a process of chemical and bromatological evaluation of some plant and animal Presidia products. They were compared to usual Italian bromatological references in IEO and INRAN tables.

In 2012, a collaboration of the SFFB, University of Turin, and University of Palermo studied the sustainability of a significant sample of European Presidia from three points of view: agri-environmental, socio-cultural, and economic. Four areas were investigated: environmental (biodiversity preservation, food production sustainability increase, economic (producer income rise, locally-driven activity development, employment increment), social (producer social role improvement, organizational skill consolidation, self-esteem increase), cultural (producer cultural identity consolidation, production area promotion).

Life Cycle Assessment of Presidia food products to determine their value utilized an approach that considered the entire supply chain: before, during, and after food production. To this end, Presidia products were evaluated from two different perspectives: nutritional and multi-criteria sustainability.

Four Italian Presidia were selected for analysis based on food category and available comparative literature: 'Lenticchia di Ustica', a lentil (legume) grown on an island of Sicily; 'Ramasin di Pagno della Val Bronda', a plum grown in southwest Piedmont; 'Manna delle Madonie', natural sweetener harvested from the bark of tree Fraxinus tree in Sicily; and 'Culatello di Zibello,' a crude ham from Parma.

2.2. A model of sustainability

The assessment of the socio-cultural, agri-environmental, and economic aspects of 47 of the 269 European Presidia was performed between 2000 and 2012. Presidia were selected according to risk of extinction, social sustainability, small-scale production, history, and culture.

Four areas were analyzed: environmental (biodiversity defense, food production sustainability improvement), economic (producers' income, locally driven activities development, employment increase), social (producers' role, organizational skills and self-esteem) and cultural (strengthening producers' cultural identity and promoting production areas).

They were clustered into three scales (socio-cultural, agri-anvironmental, and economic) on the basis of an assessment grid discussed and formulated by agronomists, sustainability experts, and producers. Each scale had 10 indicators that, when maximized, reached a score of 100. As an example, biodiversity was analyzed according to variety, processing tecnhnique, landscapes, seeds and intercropping. Soil and water analysis was based on rotation, irrigation, fertilization, organic fertilization; crop protection included pest and disease protection, natural pest and disease protection, weed control, natural weed control, post-harvest treatments, natural post-harvest treatments, certification; energy took into consideration renewable energy and packaging. Each indicator had a different measurement unit (yes/no - % of farms, % producers)

The Slow Food motto of "good, clean and fair" food, formed the basis of each scale. "Good and fair" parameters were mainly considered by the socio-cultural scale (taste, sustainability, link to local culture), "clean" parameters were included in the agri-environmental scale (risk of extinction, soil and water, energy, crop protection), and "fair" was also used by the economic scale (development, efficiency).

Total sustainability was analyzed over time, where T_0 represented the initial status of the Presidium and T_1 represented their status in 2012.

2.3. Nutritional evaluations

The main purpose of this part of the study was chemical analysis of Slow Food Presidia to understand if different production methods, both for animal and plant foods, could lead to different nutrient profiles. A few papers have, in fact, shown preliminary data as to the influence of seeds (native versus traditionally or genetically-modified), space (open versus greenhouses), and production type (small-scale versus intensive) on the final food.

Many of the nutritional components analyzed for each Presidium were common among the foods, but others differed according to food category. Table 1 lists the products and analyses performed for each.

Table 1. Components analyzed for each Presidium product considered.

Component	Lentil	Plum	Sweetener	Ham
Proteins	✓	✓	✓	✓
Carbohydrates	✓	✓	✓	✓
Sugars	✓	✓	✓	✓
Starch	✓	✓		
Polyalcohol (mannitol)			✓	
Dietary fibre	✓	✓		
Fats	✓	✓	✓	✓
Saturated fatty acids				✓
Monounsaturated fatty acids				✓
Polyunsaturated fatty acids				✓
Cholesterol				✓
Calcium	✓	✓		✓
Iron	✓	✓		✓
Phosphorous	✓	✓		✓
Magnesium	✓	✓		✓
Potassium	✓	✓		✓
Sodium	✓	✓		✓
Vitamin C		✓		
Flavouring		✓		

All nutritional analyses were performed in the Chemical Laboratory of the Chamber of Commerce in Torino, Italy over a four-year period (from 2008 till 2012). Table 2 displays the methods used for each analysis.

Table 2. Analytical method used for component analysis of Presidia tested.

Component	Analytical method
Proteins	Kjeldahl Method (ISO 1871:2009)
Sugars	HPLC (high performance liquid chromatography) Electrochemical Detection
Starch	Enzymatic-Spectrophotometric Method
Dietary fibre	Enzymatic-Gravimetric Method (AOAC 985.29)
Fats	Official methods such as Soxhlet Extraction after Acid Hydrolysis
Fatty acids*	Gas-chromatography of methyl esters of fatty acids (ISO 5508:1990 (E) + UNI EN ISO 12966-2:2011)
Cholesterol	HPLC (high performance liquid chromatography)
Minerals**	ICP-AES (Inductively coupled plasma -Atomic Emission Spectroscopy)
Vitamin C	Spectrophotometric method
Flavoring	HS-GC-MS (Head Space Gas Chromatography Mass Spectrophotometry)

* Fatty acids including saturated, monounsaturated, and polyunsaturated fatty acids.
**Minerals including calcium, iron, phosphorous, magnesium, potassium and sodium.

Following the analyses, study results were compared to bromatological values of similar foods reported in IEO and INRAN literature. The comparison was not possible for ‘Manna delle Madonie’ as its traits are not available in the literature.

3. Results.

3.1. ‘Lenticchia di Ustica’

These tiny, dark brown, tender and flavorful lentils grow on the fertile volcanic soil of the island of Ustica. They must be soaked and then cooked for 45 minutes.

The Presidium was launched in 2000 from a point of very good performance. Nevertheless, it attained excellent scores due to the impressive interest and involvement it attracted by young people, as well as its good defense of cultivation, soil, and water. By T_1, post-harvest chemical soil treatments against weevils had been replaced with cold treatments and all producers were adopting organic practices (Figure 1).

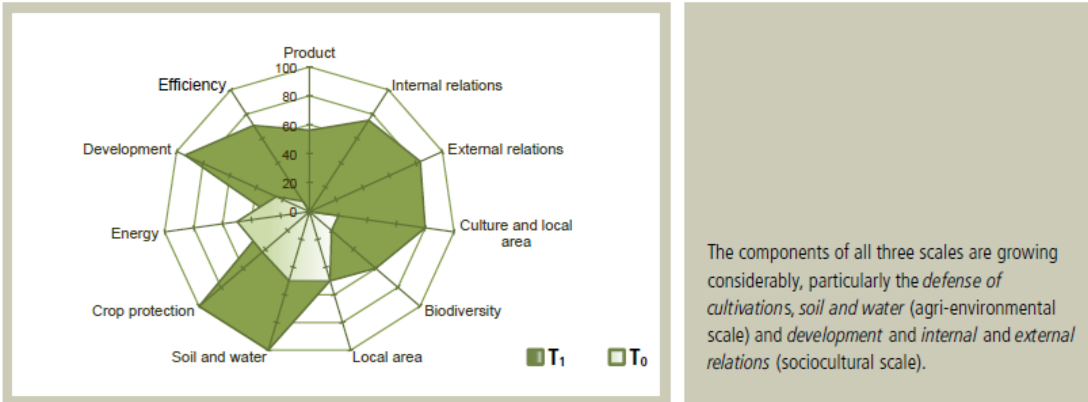

Figure 1. Sustainability performances of ‘Lenticchia di Ustica’: T_0 = initial status of Presidium, T_1 = 2012 status

The results describing the nutritional quality of the lenticchia are displayed in Figure 3. The dietary fiber of ‘Lenticchia di Ustica’ is more than double that of other lentils, which led to a reduced energy value and carbohydrate content. Furthermore, the high Iron content in the Presidium results in a single serving (about 60 g) providing 5.4 mg of iron, or almost 50% of the RDA.

Table 3. Summary of results comparing ‘Lenticchia di Ustica’ and standard lentils (per 100 grams)

	Lenticchia di Ustica	**Lentils**
Energy value (kcal)	252	291
Proteins (N x 6,25) (g)	27,7	22,7
Carbohydrates (g)	32,6	51,1
Fats (g)	1,2	1,0
Starch (g)	29,82	44,8
Sugars (g)	2,78	1,8
Dietary fiber (g)	30,22	13,8
Iron (mg)	9	5

3.2. ‘Ramasin della val Bronda’

The small, dark, and very sweet ‘Ramasin della val Bronda’ plum is familiar to many in the Piedmont (region in northwest Italy), but it is relatively unknown in the other Italian regions. It is a traditional fruit that was typically found along the borders of family orchards.

The Presidium started its activity in 2007 with a very low initial score, but the 2012 evaluations found interesting levels of efficiency and sustainability. The producers now sell their products both fresh and processed (in jam and spirits), promote them collectively, and two producers among the six are women. Organic agriculture has been only partially adopted to-date, and the packaging still requires improvement (Figure 2).

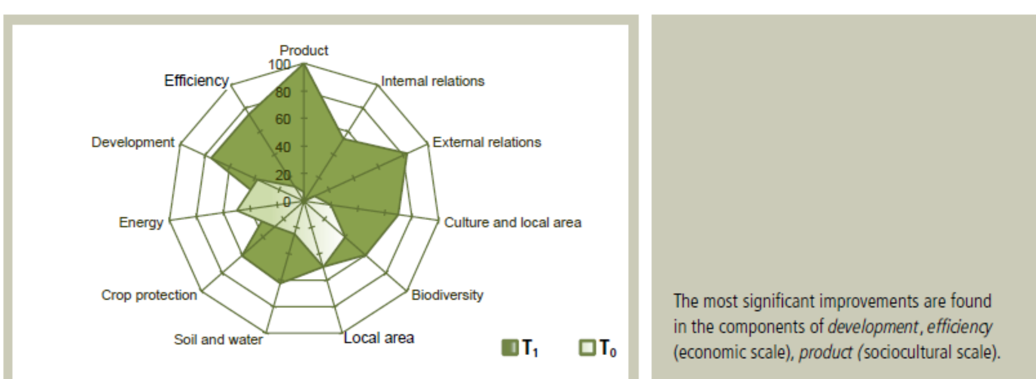

Figure 2. Sustainability performances of ‘Ramassin della Val Bronda:’ T_0 = initial status of Presidium, T_1 = 2012 status

The results of the nutritional quality analysis of the Ramasin are summarized in Table 4. The 'Ramasin di Pagno della Val Bronda' plum is richer in dietary fiber (seven-fold higher) and vitamin C (more than two-fold) compared to the reported standard values for plums. The measured flavoring components explain the organoleptic character that improves the consumption of this fruit.

Table 4. Summary of results comparing 'Ramasin della Val Bronda' and standard plum (per 100 grams)

	Ramasin della Val Bronda	**Plum**
Energy value (kcal)	41	42
Proteins (N x 6,25) (g)	0,7	0,5
Carbohydrates (g)	9,6	10,5
Fats (g)	< 0,10	0,10
Starch (g)	< 0,10	< 0,10
Sugars (g)	2,8	1,8
Dietary fiber (g)	7	1
Phorsphorus (mg)	19	14
Vitamin C (mg)	9,4	5

3.3. 'Manna delle Madonie'

'Manna' is obtained from a resinous substance extracted from the bark of ash trees in Castelbuono and Pollina (Madonie mountains, Sicily), which dries rapidly to form white tubes. It is used as a natural sweetener and has very low glucose and fructose content.

The Presidium, launched in 2002, has improved in its harvest technique. Purity and quality are now continuously monitored and the use of metal wires hung on branches to collect the manna (*versus* manna flowing down the bark) has reduced impurity levels.

Many producers are very young (seven among the 10 are under 35 years). Sustainability, already very high, has remained unchanged even with improved harvest techniques. The quantity of 'manna' produced has increased from 100 to 450 kg/year (Figure 3).

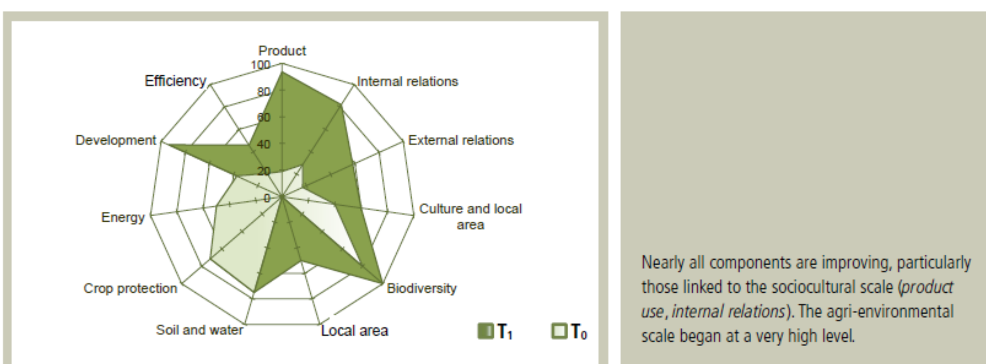

Figure 3. Sustainability performances of 'Manna delle Madonie:' T_0 = initial status of Presidium, T_1= 2012 status

The results describing nutritional quality of manna are mentioned in the Table 5. The 'Manna delle Madonie' main characteristic is its high mannitol and low soluble sugar contents which result in a reduced energy value, but conserved sweetening power.

Table 5. Summary of results 'Manna delle Madonie' (per 100 grams)

Energy value (kcal)	**176**
Proteins (N x 6,25) (g)	< 0,10
Carbohydrates (g)	64
Fats (g)	0,3
Polyalcohol (D-mannitol) (g)	53
Sugars (g)	11

3.4. 'Culatello di Zibello'

'Culatello di Zibello' is among the most highly regarded Italian cured meats due to its long and delicate processing and prized cut of pork. The heritage and richness of the cured ham is amplified by the peculiar climate of the foggy flatland from which it comes near the Po River. The right aging is a key element that has been passed on for generations. The ham embodies the history of a land, a people's tradition, and the particular climate.

In the Protected Designation of Origin (PDO) production, which is mainly industrial one, great efforts have been made in recent years to differentiate traditional producers. In the case of 'Culatello di Zibello,' it has been improved in several ways: it is GM-free, it has shown strong growth on the sociocultural scale, and it has maximized its local area score (agri-environmental) (Figure 4).

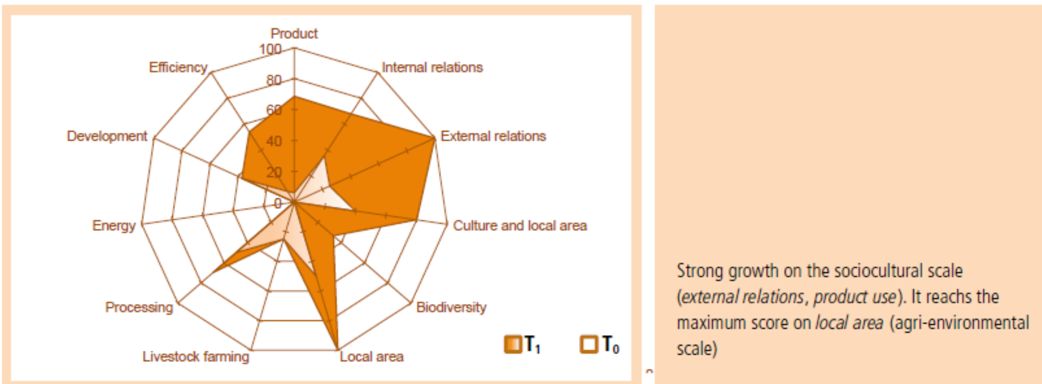

Figure 4. Sustainability performances of 'Culatello di Zibello:' T_0 = initial status of Presidium, T_1 = 2012 status

The analysis of the nutritional components of 'Culatello di Zibello' compared to other cured hams as displayed in Table 6 shows that is a protein-rich animal product. Its monounsaturated fatty acid content is higher than in other crude hams, and its cholesterol value is lower. These characteristics make 'Culatello di Zibello' very similar to low-fat bovine and ovine meat. Other positive qualities are its low sodium content and high phosphorous and potassium contents.

Table 6. Summary of results comparing 'Culatello di Zibello' and standard crude ham (per 100 grams)

	Culatello di Zibello	**Crude ham**
Energy value (kcal)	404	370
Proteins (N x 6,25) (g)	29	22
Carbohydrates (g)	< 0,10	< 0,10
Fats (g)	32	31
Saturated fatty acids (g)	11	10
Monounsaturated fatty acids (g)	16,2	13,9
Polyunsaturated fatty acids (g)	4,4	3,8
Sugars (g)	< 0,10	< 0,10
Cholesterol (mg)	72	92
Sodium (mg)	1544	2378
Phosphorous (mg)	320	177
Potassium (mg)	646	281

4. Discussion

Lifestyle and nutrition health indicators show a pandemic of obesity, type-2 diabetes, some cancers, and many chronic degenerative diseases, including Parkinson's and dementia. Given these data, it is no surprise that there is an increased consumer demand for healthier food choices. While fortified and deeply transformed foods can augment the intake of protective nutrients (fibers, antioxidants, chemo-protective substances, healthy fats, and more), their taste and other aspects often take them far from natural, traditional, and local food products. On the other hand, local and traditional certified foods often suffer from a lack nutritional and bromatological data. Furthermore, there is an increasing concern about the ecological footprint of food. How best to combine sustainability, health, and pleasure deserves consideration. Slow Food Presidia responds to this issue through its focus on several crucial sustainability issues of local communities: small-scale agriculture and farming, mountain pasture protection and pastoral farming, traditional landscape defense, traditional seed selection and propagation by communities, animal welfare, transparent labeling and ecologically sustainable packaging.

Nutritional quality analysis of Presidia confirms a strong consumer health orientation and suggests a new global 'health-oriented LCA' approach. Understandably, the approach is limited to small productions that represent a very small percentage of world food production. However, Presidia present an interesting paradigm that can be applied to other small-scale productions in order to promote LCA analysis in less-industrial settings while taking into account holistic gastronomic aspects that are sometimes forgotten in such studies.

5. Conclusions

The combined approaches represent an interesting example of Life Cycle Assessment (LCA) of food. In order to consider the global impact on the well being of both land and people, the nutritional aspect must be included alongside the usual sustainability markers.

Slow Food Presidia products (both plant and animal foods) performed at high levels of global sustainability and demonstrated extra nutritional value. Our work suggests that when a nutritional evaluation is combined with traditional LCA social, agri-environmental, and economic aspects, an innovative tool emerges that help small scale producers improve product characteristics, as well as support and promote traditional gastronomic activities. Furthermore, the food—health relationship must be considered with a multifactorial approach that encompasses producers, consumers, communities, and environment.

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Progress toward an LCA impact assessment model linking land use and malnutrition-related DALYs

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ABSTRACT
So far, land use impacts in life cycle assessment have only been considered in the context of ecosystem quality and ecosystem services. However, in a manner similar to water consumption, land use has the potential to impact food production and thereby human health. In this paper we develop a model to quantify food production losses due to land use and combine the food loss with a socio-economic, trade-based model to account for DALYs from malnutrition due to reduced food production potential in each country. Although high uncertainties occur along the cause-effect chain, a case study involving cotton production demonstrates that impacts can be quantified and are relevant in comparison to impacts associated with water consumption and other life cycle emissions. The paper provides preliminary maps of characterization factors at a high spatial resolution.

Keywords: impact assessment, land use, land occupation, land use footprint, trade-offs

1. Introduction

Land use impacts are an essential consideration when conducting life cycle assessment (LCA) in the agri-food sector. In most cases, without the inclusion of land use impacts it is unlikely that a reliable assessment of overall environmental performance can be achieved and the core principle of *comprehensiveness* (ISO 14040: 2006) satisfied. When considering alternative agri-food production systems and products, land use impacts are also central to the assessment of trade-offs, which frequently exist between land and water use (Pfister et al. 2011; Ridoutt et al. 2013; Stoessel et al. 2012), as well as other impact categories such as greenhouse gas emissions (Müller-Wenk and Brandão 2010). In addition, land use impacts are pertinent to many contemporary policy questions, such as the environmental benefits of sustainable intensification in agriculture (Garnett et al. 2013), bio-fuels (Searchinger et al. 2008), alternative protein sources (Wirsenius et al. 2010), sustainable diets (Heller et al. 2013), and land sparing/sharing strategies (Phalan et al. 2011) to name a few. The criticality of assessing land use in LCA stems from the recognition that from a global perspective land resources are already scarce (Lambin et al. 2013), many of the major global environmental challenges directly relate to land use (Rockström et al. 2009), and the increasing world population and affluence are creating unprecedented demand for food, fibre and bio-energy products.

However, at the present time, impact assessment modelling for land use in LCA is relatively immature (Hauschild et al. 2013; Jolliet et al. 2014). Although numerous land use indicators exist (Mattila et al. 2012), many closely resemble life cycle inventory parameters and others are stand-alone, proxy indicators (e.g. ecological footprint) which lack coherence with the overall life cycle impact assessment framework, making interpretation and assessment of trade-offs difficult. Part of the complication in assessing land use impacts in LCA arises from the multiplicity of cause-effect pathways leading to the various areas of protection. Recently, new frameworks and methods have been proposed for assessing biodiversity impacts (de Baan et al. 2013; Koellner et al. 2013) and ecological functions (Bare 2011; Bos et al. 2012; Saad et al. 2011). In addition, land has been considered from the perspective of a resource which has the potential to be degraded in its productive capability for future users (Garrigues et al. 2012; Núñez Pineda 2011).

Our research involves modelling the impacts of land use related to the area of protection Human Health, which has been completely neglected so far. In particular, we focus on the impact pathway linking land occupation and protein-energy malnutrition (PEM), expressed in Disability Adjusted Life Years (DALYs), taking an approach that builds on previous modelling of human health impacts related to water use (Boulay et al. 2014; Motoshita et al. 2010; 2014; Pfister et al. 2009) The model contributes essential new analytical capability for

LCA studies concerning agri-food and other land-based production systems, and is intended to ultimately be integrated with models addressing land use impacts on ecosystems and resources to enable a complete land use impact assessment capability.

2. Methods

2.1. Model structure

The model structure connecting land use to human health impacts is depicted in Figure 1. The Occupation of land leads finally to DALYs from malnutrition. Land use change was not considered to have an immediate effect on human health other than the effects related to loss of ecosystem services and biodiversity, which are the subject of the methods described above.

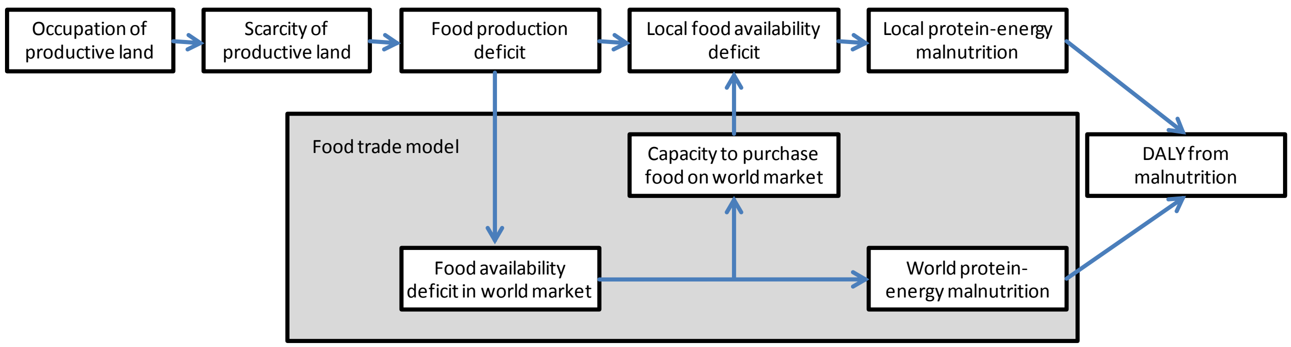

Figure 1. Land use impact pathway leading to protein-energy malnutrition

The first step was a spatially explicit characterization of land use productivity as a proxy for the relevance of a m^2yr of land in global comparison, by applying the land stress index developed in Pfister et al. (2011), which we call hereafter the Land Productivity Index (LPI). Net primary productivity of potential vegetation was used to relate each m^2yr of land occupation to m^2yr of global land unit on 5 arc minutes spatial resolution (~10km).

The subsequent steps were calculated on political units (countries). Scarcity of land within a region was determined in order to quantify the deprivation potential (i.e. the probability that land use deprives another user of land due to competition for productive land as a limited resource within a region). We applied a factor to account for the land competition within a country since some countries have declining agricultural activities and therefore land use might just replace other land use not producing food. For this we developed a Cropland Scarcity Index (CSI) in relation to the share of agricultural GDP to total GDP using data from the CIA World Factbook (CIA 2010) as shown in Figure 2. For countries with agricultural GDP shares of <1%, the CSI was set to 0.05, for shares between 1 and 4% the CSI increases linearly to 0.8, between 4 and 5% the CSI was set at 0.8, between 5 and 10% at 0.9 and for shares >10% the CSI was set to one, assuming full competition.

By combining the LPI_i (on grid cell level) with CSI on national level, the deprivation of land in absolute terms of highest land quality was obtained. This result was normalized by the production weighted average LPI_j in each country *j* derived from Pfister et al. (2011) with application of the national food loss factor (FLF_j) obtained by quantifying food energy production per m^2 yr of land use, based on FAO (2014) in order to get the food energy loss of land use in each country (MJ / m^2yr).

From here, the food loss model developed by Motoshita et al. (2014) was applied to account for the local and trade-related effects of food production loss in each country. This model reports local as well as global protein-energy caused malnutrition and is linked to DALY based on health damages occurring from malnutrition in each country leading to the effect factor for each country ($EF_{country}$) relating DALY to loss of food production (DALY/MJ).

The final characterization factor for each grid cell *i* (CF_i) in country *j* is calculated as follows:

$$CF_{LU,\ malnutrition,\ i} = LPI_i / LPI_j * CSI_j * FLF_j * EF_j \qquad \text{Eq. 1}$$

In order to account for the domestic health impacts we applied the EF for domestic malnutrition by Motoshita et al. (2014) on country level ($EF_{dom,j}$). The domestic CF ($CF_{LU,\ malnutrition,\ dom,\ i}$) is calculated as:

$$CF_{LU,\ malnutrition,\ dom,\ i} = LPI_i / LPI_j * CSI_j * FLF_j * EF_{dom,j} \qquad \text{Eq. 2}$$

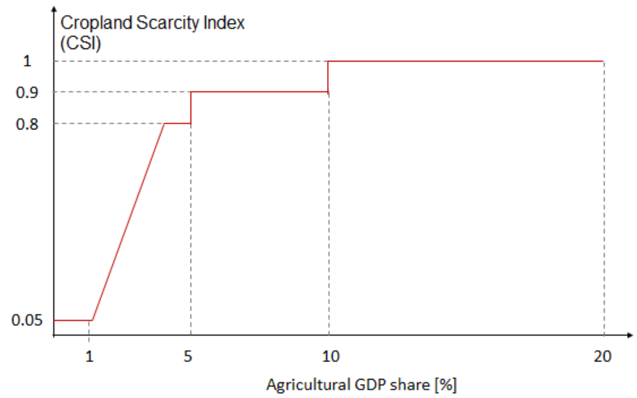

Figure 2. Cropland Scarcity Index (CSI) as a function of national GDP share of agriculture

2.2. Comparison of human health impacts caused by freshwater consumption and land use

In order to compare human health impacts from freshwater consumption and land use directly, we calculated water consumption damage equivalents for land use (WC_{eq}), which is indicating how much water consumption would cause equal damage as land use (m^3 / m^2yr). For assessment on high level of detail, the water consumption impacts on human health from Pfister et al. (2009) for each watershed *k* ($CF_{WC,\ malnutrition,\ k}$) were normalized to grid cell domestic land use impacts ($CF_{LU,\ malnutrition,\ dom,\ i}$) for each 5 arc minute grid cell ($LU_{eq.,i}$):

$$WC_{eq.,i} = CF_{LU,\ malnutrition,\ i} / CF_{WC,\ malnutrition,\ k} \qquad \text{Eq. 3}$$

The water consumption related impacts only account for domestic effects and therefore they are compared to domestic land use effects only. Furthermore, the cause-effect model by Pfister et al. (2009) leads to lower results per lack of food than other models (Boulay et al. 2014) and therefore we applied also the upper level of the 95% interval ($CF_{WC,\ malnutrition,\ 97.5\%,\ k}$, as provided by Pfister and Hellweg 2010) for comparison that accounts for sensitivity:

$$WC_{eq.,\ 97.5\%,\ i} = CF_{LU,\ malnutrition,\ i} / CF_{WC,\ malnutrition,\ 97.5\%,\ k} \qquad \text{Eq. 4}$$

2.3. Cotton case

We applied the developed factors to assess human health impacts caused by land use in cotton production based on the land use data calculated by Pfister et al. (2011) on high spatial resolution (5 arc minutes). We compared the impacts of land use to those of water use by multiplying the water consumption data of Pfister et al. (2011) with $CF_{WC,\ malnutrition,\ k}$.

3. Results

The newly developed model reports land use induced health impacts for each 5 arc minute grid cell as presented in Figure 3A. Figure 3B shows the domestic effects and reflects that industrialized countries and some emerging economies are modeled to have few or no impacts within the country but export impacts through variation in trade with other regions.

Figure 4 presents the comparison of land use and water consumption impacts on malnutrition. The damage equivalents for water consumption in terms of land use show no impacts for land use but impacts for water use in many emerging economies, which is due to the different model of health impacts in Pfister et al. (2009) compared to our approach. In many regions impacts of more than 100 m^3 correspond to 1 m^2 yr, which is a very high ratio considering that on global average 1 m^3 of rainfall occurs on 1 m^2 yr (1000mm rainfall per year). This is especially the case when comparing with the expected water consumption impacts. The upper level of the 95% interval for water consumption changes the picture quite a lot as e.g. in the case of the Indian sub-continent. This comparison already allows to estimate the relevance of water consumption and land use in agriculture. Most tropical regions (mainly except India) have relatively high volumes of water as land use equivalents and most probably land use is the higher problem. The case of India shows no robust trend in terms of priority of land or water use (compare Figure 4A and B).

The cotton case study reveals damage on human health due to land use and water consumption per tonne of harvested seed cotton (Figure 5). Cotton production causes especially high impacts in tropical regions due to land use (Figure 5A) and some significant impact due to water consumption (based on the expected value) in arid regions such as India, Egypt and the Aral Sea region (Figure 5B). To evaluate the impact of land use compared to water consumption in the cotton case, land use impacts are divided by water consumption impacts at the upper confidence level (Figure 6). In emerging economies (where domestic land use impacts are set to zero) and in arid regions water consumption impacts are higher. In India the impacts are at the same level, while in other tropical regions land use impacts are higher in most cases.

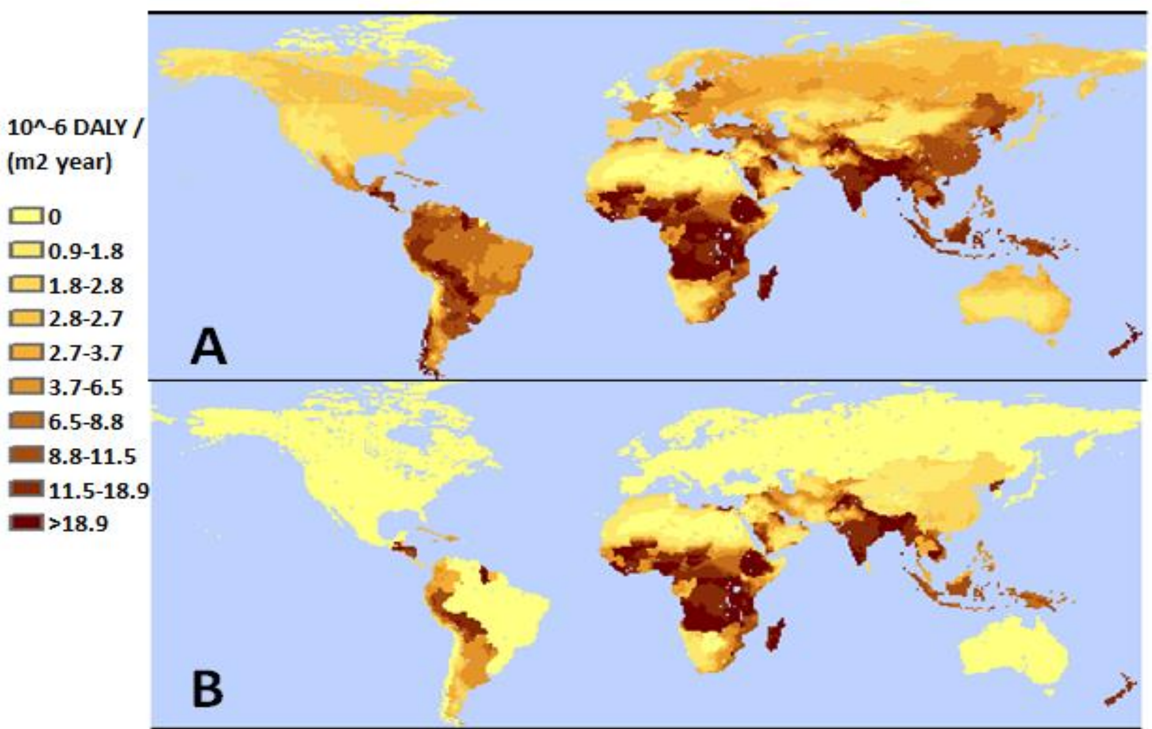

Figure 3. Characterization factors quantifying DALY from malnutrition per m^2 year of land use on 5 arc minutes spatial resolution. (A) shows total impacts including trade effects ($CF_{LU,\ malnutrition,\ i}$) and (B) only domestic impacts ($CF_{LU,\ malnutrition,\ dom,\ i}$).

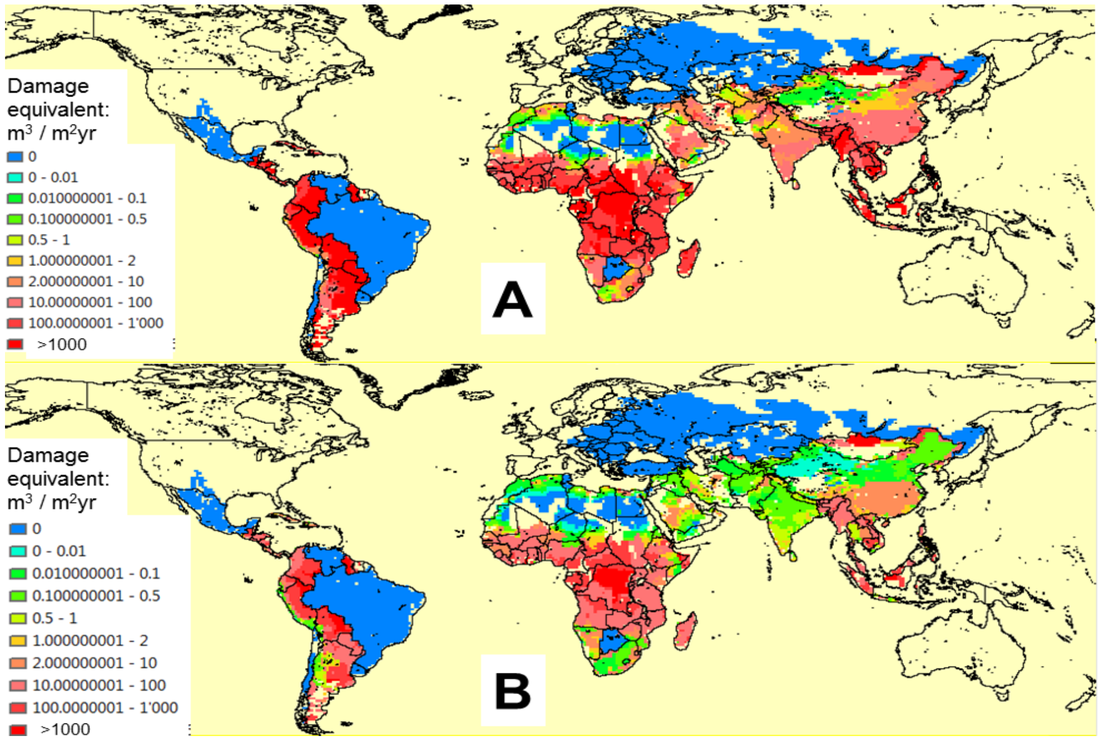

Figure 4. Comparison of human health impacts (domestic impacts only, trade-related impacts excluded) related to land use and water use. (A) Based on expected water consumption impact ($WC_{eq., i}$) and (B) based on 97.5% percentile estimate of water consumption impact ($WC_{eq ,97.5\%,,i}$)

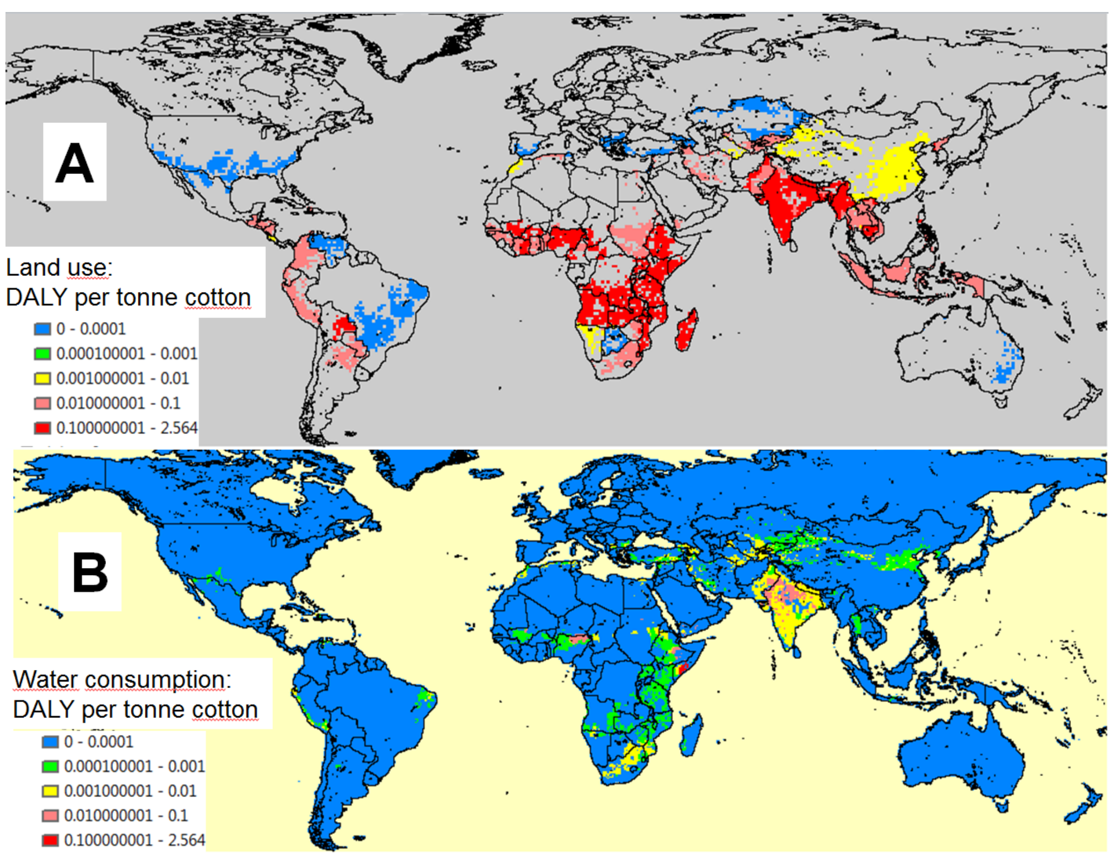

Figure 5. Cotton case study results comparing potential human health impacts from (A) land use (based on $CF_{LU, malnutrition, dom, i}$) and (B) water consumption (expected value, based on $CF_{WC, malnutrition, k}$)

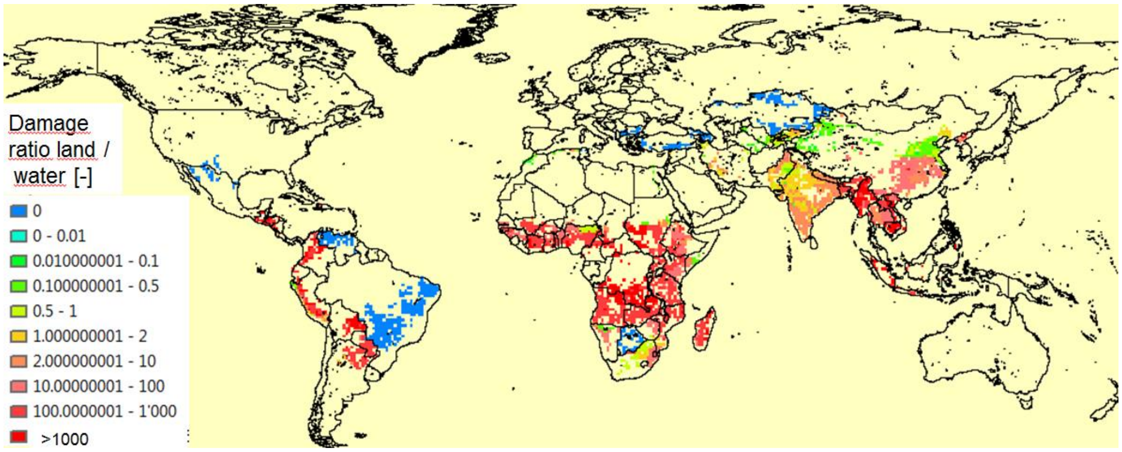

Figure 6. Ratio of impacts between land use and water consumption for cotton production. For water consumption the upper level of the 95% confidence interval (based on $CF_{WC,\ malnutrition,97.5\%,\ k}$) is used.

4. Discussion

Through the agency of trade, the local consumption of goods and services can impose pressure on land resources throughout the world. It is estimated that 197 million ha (or 33%) of the land supporting US consumption is located in other countries (Yu et al. 2013). For the EU and Japan, the figures are 50 and 92% respectively (Yu et al. 2013). Assessment of the environmental impacts of land use from a life cycle perspective is therefore critical. High levels of consumption in developed countries can be contributing to land scarcity in countries where land and water resources are already insufficient to meet the needs of local communities and access to food is limited by low incomes (Fader et al. 2013; Weinzettel et al. 2013). In addition, activities which are intended to have a positive environmental impact, such as land conservation and the cultivation of bio-fuel and energy crops to avoid the combustion of fossil fuels, could also have unintended negative consequences. Although a complete suite of LCA impact assessment models for land use addressing all Areas of Protection remains a future aspiration, the modeling presented in this paper is a first step toward filling the gap in relation to land use and human health. We also envisage this new model contributing to a land footprint indicator which goes beyond ecosystem impacts.

It might seem counterintuitive that the model can potentially be used to report land use impacts in relation to malnutrition for production systems where land is used for food production. However, in LCA environmental impacts are quantified in relation to the life cycle of a functional unit and not the direct benefit of the functional unit itself. As such, the model can be used to compare alternative food production systems and the potential impacts associated with using arable land for purposes such as textile and bio-energy production. In any case, we do not make assumptions about the way crop products are utilized: direct consumption, utilization for livestock, bio-fuel or other industrial production, waste, or contribution to excessive food consumption beyond a healthy diet, since this is part of the functional unit in LCA and not impact assessment.

The impact assessment model presented in this paper has three main components: land productivity assessment, land scarcity assessment and the food production-trade-DALY model. The current model is operational, as demonstrated by the case study presented, however we regard it as preliminary and subject to ongoing refinement, especially in relation to the first two model components. There is potential for refinement of the Land Productivity Index in terms of the relationship to NPP_0, by the inclusion of other factors which influence suitability for cropping, as well as the local level of technological advancement in agriculture. The land scarcity assessment seeks to describe the extent to which land occupation in a country constrains the agricultural sector. This assessment is based on the % GDP from agriculture at the country level. In other words, land use is deemed to have a greater impact on food production in countries where the agricultural sector is a large proportion of GDP. However, the relationship is not considered straightforward, depending in part on the potential for additional land conversion to agriculture as well as local preference for employment in the agricultural sector and numerous other factors. An improvement of the Cropland Scarcity Index could be achieved by integrating assessment of available and used land for agriculture, analysis of past trends in agricultural land use, as well as productivity of different land use types and related competition among sectors. It is important to note that in calculating potential food production deficits, care has been taken to avoid double counting due to irrigation water scarcity which is separately accounted in water use impact assessment models (as shown in the cotton case study).

Due to the high complexity of the cause effect chain and data limitation the characterization factors seem to have a very high uncertainty and might be in a similar range as those provided for water consumption (Pfister and Hellweg 2010). Quantification of uncertainties needs to be done as a future step along with improving the model components as described above.

The cotton case study shows that land use impacts on human health might cause one DALY per tonne of seed cotton in an extreme case and more than one DALY per 10 tonnes in other cases (e.g. India and Africa). In the case of India, these results would suggest around 10% of national malnutrition-related DALYs are attributable to cotton production, which occupies around 8% of arable land area. That said, the uncertainty of these numbers is even higher than the uncertainty of the CFs since the global model of land use in cotton production also has high uncertainties (Pfister et al. 2011). However, the range of the values shows relevance when compared to overall impacts of cotton production on human health, which is in the order of 0.01 DALY per tonne based on ecoinvent v2.01 data (Ecoinvent Centre 2008) and the eco-indicator 99 method (Goedkoop et al. 2001) without land and water related health impacts.

5. Conclusion

Addressing potential human health impacts caused by land use has shown to be relevant and possible. In particular, our method strengthens the ability to assess trade-offs between GHG emissions, land and water use (Pfister et al., 2011; Ridoutt et al., 2013) which are central to concerns about sustainable food production and sustainable diets. The method will also be vital to exploring the impact of land use systems that compete with food production (e.g. bio-fuels) as well as the role of livestock products in a sustainable global food system since livestock systems vary enormously in feed supply (and therefore productive land requirements) and have the capacity to both add to or subtract from the global food system (Foley et al., 2011).

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Trade-offs between agricultural product carbon footprints and land use: a case study from Tanzania

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ABSTRACT

A case study comparing extensive, low-yielding smallholder maize production with an intensified, high-yielding production system ('sustainable intensification') is used to illustrate potential trade-offs between agricultural product carbon footprints (PCFs) and land use (LU). In the comparative analysis the extensive systems had lower PCFs so the conclusion may be that these systems should be encouraged as a climate mitigation strategy. However, if LU and the potential for land use change as a consequence of the extensive systems' low yields are also considered, then the conclusion changes in favour of the intensified, high-yielding system. Our conclusion is that an assessment of the success of sustainable intensification with regard to climate mitigation needs to also include indirect effects at a larger spatial scale than individual farms in order to avoid misinterpreting PCF results.

Keywords: sustainable intensification, maize, greenhouse gas emissions, extensive farming, land use change

1. Introduction

The agricultural sector is key for addressing various interconnected challenges, including food security, land use and climate change. Significant yield reductions are predicted over the next decades in many regions of the world as a result of climate change (IPCC 2007). At the same time, agricultural activities contribute to global warming by releasing greenhouse gases (GHGs) at all stages of agricultural value chains. The application of sustainable practices along entire food supply chains is essential for reducing GHG emissions, ensuring continued agricultural production, increasing global food security and feeding the growing world population (Beddington et al. 2012). The concept of sustainable intensification, i.e. the production of more food from the same area of land while reducing associated environmental impacts, is of particular importance in this respect (Beddington et al. 2012).

Product carbon footprints (PCFs) estimate the amount of GHGs emitted during the life cycle of goods and services. They are reported per unit of product and aim at maximising input-output ratios, where high input-high output systems can have low PCFs, i.e. low levels of GHG emissions per unit of output. Despite the importance of GHG accounting per unit of product for assessing efficiencies and potential leakage effects, such analyzes are lacking for many agricultural production systems and agroecological and socio-economic contexts (Smith and Wollenberg 2012).

Our case study on the PCF of maize cultivation in smallholder production systems in Tanzania is part of a project investigating sustainable intensification practices. So far, little information has been reported on the GHG emissions from smallholder production as previous analyzes have focused on large export oriented farms (e.g. Edwards-Jones et al. 2009). Our analysis includes extensive, low-yielding farming systems as practiced widely in the region, and an improved, more intensive management system which aims at increasing yields by applying more targeted inputs. The PCF of these two systems are compared and their respective climate impacts discussed. The hypothesis is that raising yields by applying more targeted inputs under the concept of sustainable intensification can reduce the climate impact from maize production.

2. Methods

2.1. PCF calculation

For the calculation of farm gate PCFs we used data on smallholder maize production in Tanzania generated as part of a larger public-private partnership project implemented by the Norwegian University of Life Sciences, Sokoine University of Agriculture in Tanzania, Syngenta and Yara International. Field trials were conducted on five smallholder farms located in the Njombe and Morogoro regions of Tanzania. At each site, paired treatments were established where the farmers continue with their traditional management ('farmer practice', FP treatment) on one plot. On a second plot more targeted farm inputs are applied, following a protocol devised by the project partners ('YSS treatment'). At all sites soil samples were taken at the beginning of the trials in order to develop a balanced and crop-specific nutrition program.

The system boundary of the PCF analysis included the production and transport of farm inputs and all relevant on-farm processes up to the farm gate for one project year. Primary data on the amounts of fertilizers, agrochemicals, yields and crop residues were available for each treatment. Direct N_2O emissions from soils as a result of nitrogen inputs (mineral fertilizers, crop residues) were estimated using the IPCC (2006) Tier 1 method. Indirect N_2O emissions were calculated using the Bouwman et al. (2002) model for ammonia volatilisation and a nitrogen balance approach to identify potential nitrate leaching losses. Mineral NPK fertilizers used in the YSS treatment were produced in Norway and transported to Dar es Salaam by container ship. Mineral fertilizers applied on FP plots were assumed to be produced in China and Morocco and shipped by bulk. Emission factors for the production of inputs were obtained from Yara International, Brentrup and Pallière (2008), IFA (2009), Saling and Kölsch (2008) and PE International's GaBi 4 database.

2.2. Description of the case study farms

All field operations on the case study farms were carried out manually. The YSS treatments received 138 kg N ha^{-1} $year^{-1}$, 57 kg P_2O_5 ha^{-1} $year^{-1}$, 34 kg K_2O ha^{-1} $year^{-1}$ and micronutrient applications. On two of the FP plots, no fertilizer was applied at all whereas some nitrogen (11-79 kg N ha^{-1} $year^{-1}$) and phosphorus (28-57 kg P_2O_5 ha^{-1} $year^{-1}$) was applied on the other three FP plots. Weed control was mainly done manually on the FP plots whereas plant protection agents were applied on the YSS plots as and when necessary. Grain yields achieved on the YSS plots ranged between 2.1 and 6.3 t ha^{-1}. Grain yields on FP plots were considerably lower (1.8-4.0 t ha^{-1}). Stover yields were 1-5 times greater on most YSS plots, leading to a greater return of organic residues to the soil.

2.3. Land use (LU)

A second indicator, land use (LU), was calculated to compare the paired treatments and highlight trade-offs between PCFs and LU. We define LU as the area of land required to produce a unit of output (Tuomisto et al. 2012), i.e. as the inverse of the yield.

2.4. Potential land use change (pLUC)

If low-yielding maize production systems as in our FP treatment are expanded to meet growing demands for Tanzania's rapidly increasing population (UN 2010), this will cause GHG emissions outside of the farms studied due to land use change (LUC). Therefore, it is not direct LUC (dLUC) taking place on the farms analyzed. Although it is an indirect effect of the extensive farming system outside of the PCF system boundary, we suggest not to calculate these indirect impacts using established methods for estimating indirect LUC (iLUC) emissions. Such iLUC occurs outside of the product system being assessed as a result of the conversion of land due to changes in agricultural land use elsewhere in the world. Modelling iLUC impacts is complex and based on assumptions about economics, market factors, pre-conversion land use types and other factors (Prins et al. 2010). In our case study region, in contrast, it is likely that LUC will be induced by an expansion of extensive farming to increase production under a business-as-usual scenario (FAO 2012). This LUC occurs nearby in order to meet growing demands for the same crop in the same region. We call this form of land use change 'potential LUC' (pLUC) and estimate associated GHG emissions based on the available land use type in the region for an extended FP system, FP_{ext} (Brentrup and Pallière 2008). First, we calculate the amount of land that needs to be converted to produce the same amount of maize harvested in YSS under the current FP management. This additional area of land is then assumed to be subject to LUC, leading to the emission of 12.4 t CO_2e ha^{-1} and year (IPCC 2006, Tier 1) if emissions are allocated evenly across 20 years (BSI 2011). In the case study area tropical shrubland is potentially available to support this expansion. These emissions, scaled by the yield on each FP plot, are added to the FP PCF for the amount of land that needs to be converted to match the YSS yield.

3. Results

3.1. Farm gate PCF

The farm gate PCF of maize varied greatly between 102 and 963 kg CO_2e t^{-1} grain. A comparison of the results for the paired YSS and FP systems reveals that in most cases the FP systems had lower PCFs (Fig. 1a). The main emissions sources on all farms were related to fertilization and crop residues. On the YSS plots, the production and transport of mineral fertilizers to the farm, direct N_2O emissions from soils, and N_2O emissions from crop residue management were the main emissions sources. On the FP sites, N_2O from crop residues was the main source of emissions with the exception of Farms A and B, where the production and transport of mineral N fertilizer was the greatest contributor, closely followed by direct N_2O emissions from soils. On the two FP plots that did not receive any mineral fertilization (Farms D and E), the only sources of emissions were N_2O from crop residue management and the production of seeds.

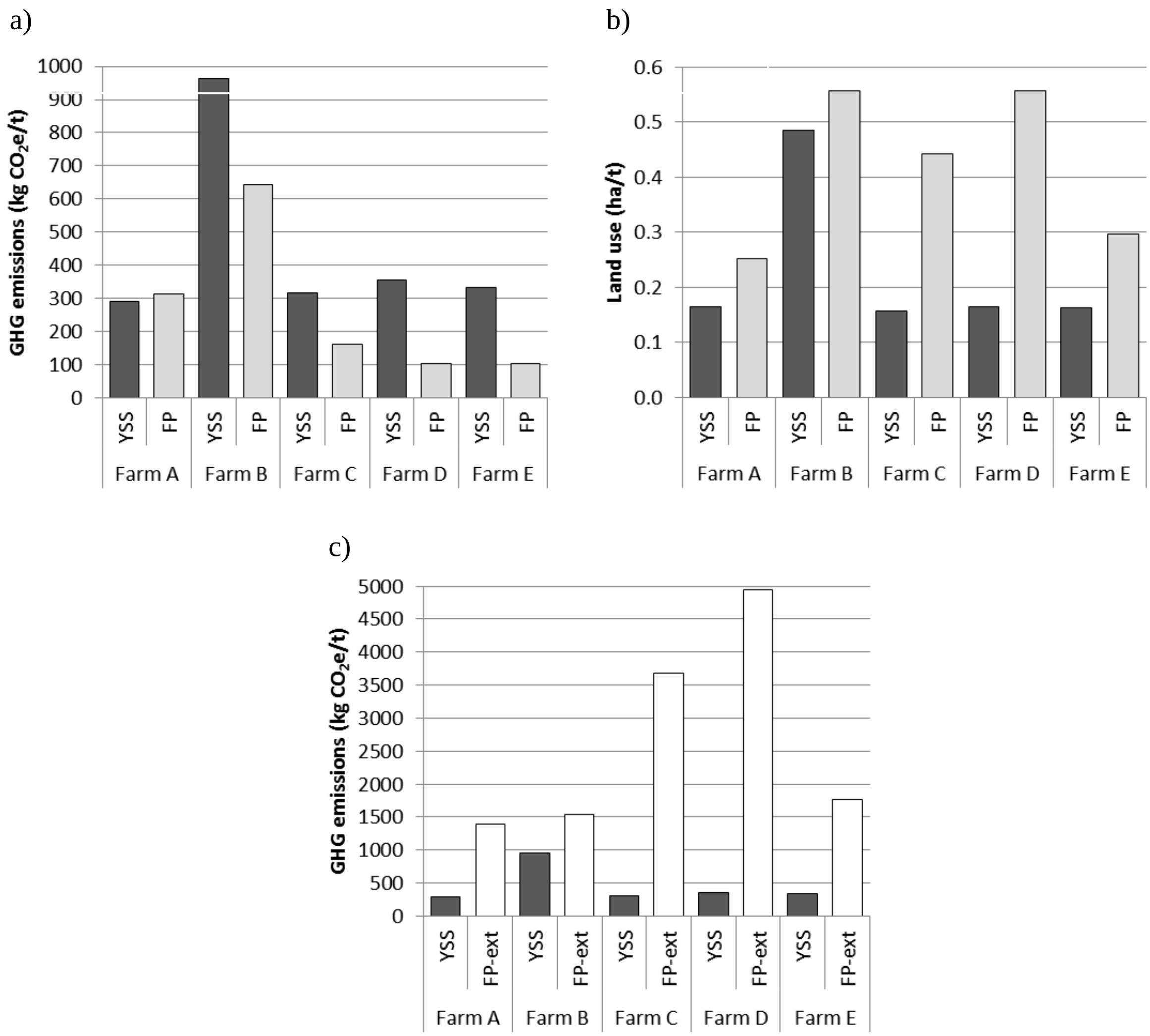

Figure 1. Results for maize production at five sites of paired management treatments: a) Product carbon footprint (PCF) up to the farm gate; b) Land use; and c) PCF including potential land use change. The black bars denote the intensified YSS management system, the grey bars the widespread farmer practice (FP) and the white bars the expanded FP system that matches YSS yields, including potential land use change on the additional land needed to produce this yield (FP_{ext}).

3.2. Land use

The LU of the case study farms ranged between 0.16 ha t^{-1} and 0.56 ha t^{-1} (Fig. 1b). Contrary to the PCF results where the YSS farms generally were associated with greater GHG emissions per t of maize than the FP farms with their extensive management, the LU indicator shows the exactly opposite pattern: for all pairs of YSS vs. FP management LU is lower and therefore more favourable for the YSS systems. The transition from FP to YSS thus indicates a move towards a lower LU and a more efficient use of land. In order to produce one ton of maize, the FP plots needed between 1.1 and 3.8 times more land than the paired YSS plots.

3.3. Indirect impacts from land use change

If GHG emissions arising from FP_{ext} from an area large enough to match the paired YSS yields (including pLUC for the additional area) are considered, the conclusions drawn from considering the farm gate PCF only change dramatically (Fig. 1c), with GHG emissions related to FP_{ext} between 2.4 and 48 times greater than the FP PCFs up to the farm gate.

4. Discussion

4.1. Smallholder maize production

The present study demonstrates the significant potential for achieving yield increases in smallholder systems by addressing limitations to productivity and applying targeted inputs of fertilizers and agro-chemicals. The wide range in PCFs between farms and within paired treatments also shows the impact of individual farm management decisions and location specific conditions such as soil pH or soil nutrient supply. Nutrient deficits and soil degradation which limited yields on smallholder farms were identified and improved, leading to significantly increased yields in the YSS system. For most FP systems, nutrient mining was observed which explains their low yields but also shows that these systems are not sustainable in the long run without increasing the input of plant nutrients in the form of organic or mineral fertilizers.

4.2. Trade-offs between PCFs and land use

If LU is considered, the conclusions differ from those based on the farm gate PCF where the extensive FP systems which use very little inputs might be taken for more environmentally friendly. When looking at the land area needed to produce a ton of maize, the FP plots clearly exert a much greater pressure on land resources than the intensified systems, which is likely to lead to land use change in the region to satisfy growing demands. This trade-off can lead to significant indirect impacts beyond the farm gate which are important to consider before concluding on the respective climate friendliness of the contrasted systems. LU can draw attention to this trade-off and should be considered whenever systems with great yield differences are compared. Out of two farms or production systems with similar PCFs but a high LU for one and a low LU for the other, the latter should be preferred.

The UK Royal Society (2009) defined sustainable intensification as “the production of more food on a sustainable basis with minimal use of additional land”, clearly recognising the need to reduce LUC and the continuing expansion of agricultural areas while meeting the challenge of increasing global food production. Our study illustrates the potential climate change impact of expanding the present low-yielding maize systems. The results emphasize the need for meeting increasing food demands without further LUC by carefully intensifying current production systems. The analysis of pLUC also highlights the need for more integrated assessments and policies, e.g. on land use and climate change, and a wider landscape scale approach to assessing the environmental impacts of farming systems and comparing the climate impact of different production systems. This is particularly important in regions where low-yielding systems dominate and increasing demands are likely to be met by converting land locally (pLUC).

5. Conclusion

The present case study highlights the risks associated with the interpretation of single issue indicators such as PCFs. In contrast, in a full LCA, trade-offs within the same system boundary become evident (e.g. Tuomisto et al. 2012). Our results show that raising the productivity and long-term sustainability of the systems analyzed by applying more targeted inputs and addressing nutrient limitations can result in a greater climate impact than the current low-yielding systems within the system boundary of a farm gate PCF. However, the benefits of sustainable intensification should be assessed at a greater spatial scale. In order to achieve food security for current and future populations, large increases in food production are necessary. Against this background, an assessment of the climate impact of extensive vs. intensive agricultural systems with significant differences in yield levels should consider trade-offs between PCFs, yields and LU. In such cases it is important to consider indirect impacts that may occur beyond the farm gate before concluding on the climate friendliness of one or the other system. A comparison of the LU indicators of the different systems can highlight the risk of LUC and help interpret PCF results. If this is not done, there is a risk that PCF results may be misinterpreted and agricultural systems that have low PCFs but exert a large pressure on land resources may be encouraged, unintentionally causing significant carbon emissions due to land conversion to agriculture.

In conclusion, the success and benefits of sustainable intensification on existing agricultural land should not be assessed based on direct impacts only. There is a need to also consider indirect impacts and larger spatial scales and to develop appropriate policy responses to ensure positive results (Garnett et al. 2013). In our analysis of the climate mitigation potential of two agricultural systems a consideration of indirect land use effects can help resolve potential conflicts between climate impacts on farm and in the wider landscape, with the benefit of sustainable intensification becoming obvious at a larger spatial scale.

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Facility arrangements, food safety, and the environmental performance of disposable and reusable cups

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ABSTRACT

Conventional disposable cups, made of fossil-based plastic or paper with inner lining of fossil-based plastic, are typically associated with an unnecessary use of scarce resources and a superfluous production of waste. An alternative has become available in disposable cups from bio-based and biodegradable materials, so-called biocups, made from bioplastic or paper with inner lining of bioplastic. Many stakeholders consider disposable biocups as more environmental friendly than fossil-based disposable cups, though other stakeholders prefer reusable cups over disposable cups. Existing LCA studies show inconsistent and sometimes conflicting results, due to differences in used data and modeling choices, for LCA studies comparing different disposable cups and/or comparing disposable with reusable cups. This paper summarizes an LCA deliberately applying multiple inventory data sets and crediting principles for recycling in comparing disposable PolyStyrene (PS) cups with (1) disposable biocups of PolyLactic Acid (PLA) and paper lined with bioplastic, (2) handwashed and dishwashed reusable cups.

Keywords: Facility arrangements & food safety, Environmental LCA, Disposable and reusable cups, Multiple inventory data sets and modeling choices, Scenarios

1. Introduction

Public provision of drinking water was in the start of the previous century usually facilitated with a “common drinking cup” (as they were referred to in that time). Such “common drinking cup” could be found near water vessels and fontains in public places as parks, trains and railroad stations, department stores, schools, theatres and offices. Even hospitals made use of the “common drinking cup”, despite public awareness and scientific evidence about their role in distributing contagious diseases. Alvin Davison, professor of biology in Lafayette College (Easton, Pennsylvania, United States of America), published in 1908 an influential study “Death in school drinking cups”. This study documents the human cells and pathogens on a cup having been used nine days in row on a school, and contributed to the introduction of a disposable paper cup to curb pathogens growth and spread by the “common drinking cup”. Kansas was in 1909 the first American state to abolish by law the “common drinking cup”, and was eventually followed by all other American states putting into force similar laws (Anonymous 1995; Davidson 1991; Montreal Gazette 1908; Reading Eagle 1909).

A “common drinking cup” has meanwhile become an antiquity, and the use of disposable cups penetrated all sectors in society. Official numbers are not publicly available, but Wikiversity (2014) claims a worldwide use of 300 billion disposable cups per year (i.e. 300E9 disposable cups/year). Many restaurants and kiosks sell beverages in disposable cups for on-the-go consumption (e.g. by commuters, shopping public, or beach visitors). Disposable cups are also typically employed where absence of cleaning facilities and large numbers of customers in short time intervals make reusable cup service practically impossible. This is not only at stake for large public events like festivals and manifestations, but as well in medium and large organizations as schools and universities with peak-consumption during breaks. Disposable cups are also increasingly used in office-type organizations, typically in combination with vending-machines, to save time and money and to streamline their hot beverage facilities. Conventional disposable cup are made from fossil-based plastic (e.g. polystyrene, polypropylene, polyethylene terephthalate), or from paper with an inner lining of fossil-based plastic (e.g. polyethylene) or wax.

Whereas disposable cups were in the first half of the last century praised for their contribution to public health, the first commercial ones were even named ‘health cups’, they became in the second half of previous century increasingly associated with an unnecessary use of scarce resources and a superfluous production of waste (Butijn et al. 2014; Reinink et al. 1991). The debate about disposable cups already goes on for decennia, though the proposed solutions have slightly shifted over the years, also given changing facility arrangements for providing beverages. Reusable cups were often put forward as the obvious alternative for disposable cups in the nearby past when organizations typically had restaurant facilities or a room-to-room-service by a ‘coffee-lady’

(Reinink et al. 1991). Restaurant facilities and 'coffee-ladies' will not be easily found anymore, however, as many organizations nowadays make use of vending machines for providing hot and cold beverages. Reusable cups are also not practical, or simply infeasible, in situations where many customers need to be served in a short time. An alternative for the conventional disposable cups, however, has recently become available with the introduction of the so-called biocups. Disposable biocups are made from materials that are renewable and have the compostability label (EN 13432). The most common disposable bioplastic cups are made from PolyLactic Acid (PLA), typically produced from corn. Disposable biopaper cups are obviously from paper, but lined with a bioplastic instead of a fossil-based plastic.

The renewable and compostable characteristics make biocups in the eyes of many stakeholders more environmental friendly than the conventional disposable cups from fossil-based plastic or paper lined with fossil-based plastic (Jager 2008). Many restaurant facilities and catering services, or organizations buying these facilities or services, therefore consider a transition from conventional disposable cups to disposable biocups. Wageningen University & Research (Wageningen UR) already (partly) made this transition by replacing conventional disposable cups by disposable biopaper cups in their office-buildings having vending machines for hot beverages without automatic cup supply. The education buildings of Wageningen UR have hot beverage vending machines with automatic cup supply. Disposable biopaper cups tend to disrupt the vending machines with automatic cup supply (Butijn et al. 2014). Technology has just become available for producing disposable bioplastic cups for hot beverages, i.e. from thermo-resistant PLA, but these thermo-resistant disposable cups are not yet taken into commercial production. Cup producers wait for a sufficient large market demand, whereas potential customers wait for an actual market supply (a clumsy impasse; Potting 2013). Disposable PLA cups for cold beverages are already longer on the market.

The transition to disposable biopaper cups for vending machines with hot beverages in their office-buildings was first decided by Wageningen UR after careful consideration of the pros and cons of the earlier used disposable PS cup in comparison with disposable biocups from PLA and from biopaper (i.e. paper lined with bioplastic). The pros and cons of the three disposable cups were investigated by research employees and students of Wageningen UR in a comprehensive internal research project looking into environmental, economic as well as social aspects. It goes too far to describe the whole research project here (see therefore Potting 2013; in Dutch), but the in-depth comparative LCA study for the three disposable cups is summarized in this paper (see for details Van der Harst and Potting 2013; Van der Harst and Potting 2014; Van der Harst et al. 2014). A survey under employees and students of Wageningen UR turned out that over half of the office-building inhabitants for environmental reasons is using an own reusable cup (Butijn et al. 2014). The LCA results for the disposable PS cup were therefore also compared with the results of an additional (screening) LCA study for reusable cups washed by hand and washed in an energy-efficient dishwasher. The comparisons are summarized in this paper and their results are discussed in relation to environmental beneficial facility arrangements.

2. Methods and means

Existing LCA studies show inconsistent, sometimes conflicting results for comparisons of different disposable cups (Van der Harst and Potting 2013), and also for comparisons between disposable and reusable cups. This can be traced back to differences across LCA studies in data used, and modeling choices made. Van der Harst and Potting (2014) and Van der Harst et al. (2014) therefore deliberately applied multiple inventory data sets and crediting principles for recycling, i.e. a modeling choice, in an LCA study comparing disposable fossil-based PS cups with disposable biocups from the bioplastic PLA and from biopaper (i.e. paper lined with bioplastic). The additional LCA study comparing the disposable PS cup with handwashed and dishwashed reusable cups, in Potting (2013), refrained from using multiple inventory data sets and modeling choices for the reusable cup LCAs. The LCAs for the handwashed and dishwashed reusable cups took a screening approach. Both comparative LCA studies thus followed on this point different methodological approaches in their inventory phase, that therefore are separately described in more detail (i.e. Section 2.1 and 2.2). Both LCA studies basically took a similar methodological approach in the other LCA phases than the inventory phase (Potting 2013; Van der Harst and Potting 2014; Van der Harst et al. 2104):

Functional unit: Both LCA studies used the same functional unit of facilitating the serving of one hot beverage from a vending machine as frequently used in the Netherlands in big organizations. This functional unit puts constraints on the disposable cups, as most hot beverage vending machines with automatic cup supply in the

Netherlands use disposable cups with a volume of 180 ml (see 2.1 and 2.2 for details about the cups weights used in our LCA comparisons).

Impact assessment: Results from the inventory phase were translated into environmental impact by means of the CML Baseline 2001 methodology (Guinée et al. 2002), supplemented with the Cumulated Energy Demand (CED) from Frischknecht et al. (2003). Both comparisons therewith covered all together eleven environmental impact categories (see Table 2). Impact results were not normalized and neither weighted. ISO 14044 (2006) rejects normalization and weighting in comparative LCAs which results are to be disclosed to the public.

Interpretation: The results for the in-depth LCA study of the disposable cups were carefully evaluated against the background of the methodological approach used, quality of data obtained, and relevance of the results in the context of Dutch environmental policies. The screening LCAs of the handwashed and dishwashed reusable cups provided indicative impact results that were compared with average impact results of the in-depth LCA for the disposable PS cup. The results of both comparative LCA studies were evaluated with regard to their relevance for environmental beneficial management options.

Software: All LCAs for the disposable and reusable cups were performed in SimaPro 7.3, but impact results for the disposable cups were imported in Microsoft Excel 2010 for calculating average impact results and spread related to the applied multiple inventory data sets and crediting principles for recycling in the cup life cycles. Microsoft Excel 2010 was used in both comparative LCA studies for making the appropriate graphical representations of results.

2.1. Comparison of disposable PS cups and disposable biocups from PLA and biopaper

Van der Harst and Potting (2013) recently made a critical comparison of ten existing LCA studies comparing disposable beverage cups. These ten LCA studies only shared climate change as a common impact category. The variation in climate change results across LCA studies for each disposable cup material was quantitatively explored by Van der Harst and Potting (2013). They calculated the ratio between the highest and lowest climate change value, and found a ratio of 1.7 for PLA cups, 3.4 for petro-plastic cups, and 20 for paper cups. Since there was also no consistency among the LCAs about the cup with the smallest climate change impact, Van der Harst and Potting (2013) next qualitatively compared the data used and methodological approach followed in each of the selected ten LCA studies. Identified possible sources for the variation in outcomes were differences in the properties of the disposable cups (e.g., material choice and weight), differences in the data used, and different choices made in modeling production processes, energy production (e.g., fossil or renewable sources), and waste treatment (e.g. different allocation/crediting principles and waste treatment processes applied).

The critical comparison of the ten LCA studies was the basis for a new in-depth comparative LCA study of disposable cups. Van der Harst and Potting (2014) and Van der Harst et al. (2014) deliberately applied multiple inventory data sets and crediting principles for recycling, a modeling choice, in their LCA study comparing disposable fossil-based PS cups with disposable biobased cups from PLA and biopaper (i.e. paper lined with bioplastic). Their use of multiple inventory data sets and crediting principles for recycling involved two LCA iterations according to the following procedure (Van der Harst and Potting 2014; Van der Harst et al. 2014):

1. Initial LCAs with one inventory data set for each process in the life cycle of the disposable cups from the three selected materials (these initial LCAs used incineration as waste treatment process)
2. Contribution and sensitivity analysis to identify processes with major influence on the initial LCA results
3. Collecting additional multiple inventory data sets for all processes with an influential contribution
4. Applying multiple data sets, multiple modelling choices:
    - Applying the collected multiple inventory data sets in next LCAs,
    - Combined with one waste treatment processes (i.e. incineration, recycling, composting, or anaerobic digestion), and with
    - Applying multiple crediting principles for the disposable cup life cycles with recycling (e.g. multiple crediting principles for recycled material)
5. Calculating and presenting average impact results and their spread (highest and lowest value) for each life cycle process based on the multiple inventory data sets and crediting principles for recycling
6. Calculating and presenting average impact results and their spread (highest and lowest value) for each of ten disposable cup life cycles

Steps 1 to 6 were gone through for each of ten disposable cup life cycles. These ten disposable cup life cycles resulted from three disposable materials (i.e. fossil-based PS, biobased PLA, and paper lined with PLA), and four waste treatment processes (i.e. incineration, recycling, composting and anaerobic digestion; the latter two not being relevant for PS). This led to ten disposable cup life cycles for which multiple inventory data sets were collected, whereof three disposable cup life cycles with recycling for which multiple crediting principles were applied (see Table 1). The applied multiple inventory data sets and crediting principles for recycling are related to disposable cups as commonly used in hot beverage vending machines in the Netherlands (i.e. not necessarily for similar disposable cups used abroad). These disposable cups typically have a volume of 180 ml. Representative cup weights related to this volume were used in the analysis (see Table 1).

Table 1. Overview of the disposable cup materials, cup-weight for each material, life cycle processes and number of inventory data sets and crediting principles included in the in-depth LCA study comparing disposable cups. The LCA study covers altogether ten disposable cup life cycles, indicated by the grey-shaded cells, resulting from three disposable cup materials and four waste treatment processes (composting and anaerobic digesting are not relevant for PS). The dark grey-shaded cells indicate the three initial LCAs, each having incineration as waste treatment process and covering the rest of the life cycle for only one disposable cup material. All processes for which only one inventory data set was used, showed to be of minor importance in the contribution and sensitivity analysis.

Life cycle processes	**PS** 4.2 gram	**PLA** 4.2 gram	**Biopaper** 5.6 gram
Cradle to disposable cup material production	**3**	**5**	**5**
Transport of disposable cup material to cup manufacturer	1	3	1
Disposable cup manufacturing	**5**	**5**	**3**
Cradle to grave for the packaging of disposable cups	1	1	1
Transport of disposable cup to customer	1	1	1
Transport of used disposable cups to waste treatment	1	1	1
Waste treatment:			
- Incineration	**4**	**4**	**5**
- Recycling (recycling process/crediting principle)	**5/4**	**5/4**	**3**
- Composting		**4**	**4**
- Anaerobic digestion		**3**	**4**

2.2. Comparison of disposable PS cups with handwashed and dishwashed reusable cups (Potting et al. 2013; Van der Harst and Potting 2014; Van der Harst 2014)

The average impact results for the disposable PS cup life cycle with incineration from the in-depth LCA study was used for comparison with two reusable cup life cycles, one with handwashing and one with dishwashing of the reusable cup after use. The comparison took a one-time use of the disposable cup before disposing it, but looked into an increasing number of reuses of the reusable cup before dishwashing or handwashing. Screening LCAs were performed, i.e. screening inventory data were used, to calculate the impact results for the two reusable cup life cycles. The weight of the reusable cup was in both LCAs taken to be 370 grams, based on a random sample of reusable cups used in the Netherlands. The composition of the reusable cup was taken from Bramberg et al. (2011). The reusable cup was assumed to endure on average 1750 consumptions before breaking, and just like the disposable PS cup to go to the incinerator as waste (Hoeboer 2012).

Dishwashing: The composition of the dishwasher was taken from Kok et al. (1996), the energy use for assembling the dishwasher from Boustani et al. (2010), who also gave basis to the assumption of 2150 dishwashing-turns before disposing the dishwasher. Waste treatment for the dishwasher was ignored, which is a worst case approach as large parts of the dishwasher are probably recycled (leading to lower impact results for the dishwasher sub-life cycle). Based on currently common dishwashers from AEG and Bosch (Hoeboer 212), the dishwasher was taken to use 9.25 liters of water, 1 kWh electricity, and 1 gram of salt per washing turn. The composition and use of soap, 9.8 grams per washing-turn, was based on Bramberg et al. (2011).

Handwashing: Handwashing of the cups was assumed to be a single item-washing (as common for people using an own reusable instead of disposable cup). The use of hot water for handwashing was set on 1 liter, and the energy use for heating the water was set at 0.222MJ (heat from natural gas) as based on Eclectsite (2013). A use of 1 gram of soap per handwashing was assumed. The soap composition is based on data from the Dutch association of detergent manufacturers (2012). We assumed the use of two towels for drying one cleaned cup (Tork 2006, Jacobs 2006).

All cradle to product and electricity data in both reusable cup life cycles were taken from EcoInvent Centre (2010).

3. Results and discussion

The two LCA studies provided a wealth of information that is summarized in a number of subsections below. Detailed results and discussions can be found in the earlier publications for both LCA studies. The LCA study comparing disposable cups is addressed in Potting (2013), Van der Harst and Potting (2014), and Van der Harst et al. (2014). The LCA study comparing the PS disposable cup with dishwashed and handwashed reusable cups can be found in Potting (2013).

3.1. Disposable PS cups not better or worse than biocups from PLA and biopaper (Potting 2013; Van der Harst and Potting 2014; Van der Harst et al. 2014)

Table 2 summarizes the comparison of, i.e. the ratio between, the average impact results for all ten disposable cup life cycles with the average impact results for the disposable PS cup life cycle with incineration. The other nine disposable cup life cycles consist of one for PS with recycling, and eight for the two biocup materials (i.e. PLA and biopaper (i.e. paper lined with bioplastic) with one of the four waste treatment processes (i.e. incineration, recycling, composting and anaerobic digestion). The comparison in Table 2 may mistakenly lead to the wrong conclusion that the disposable PS cup life cycle with incineration tends to perform better than the other nine disposable cup life cycles, i.e. for PS with recycling and the two biocup materials regardless their waste treatment process. The table does not indicate, however, the considerable and overlapping spread around the average impact result for all ten cup life cycles in most impact categories as caused by applying multiple inventory data sets (and crediting principles for recycling). This large and overlapping spread in impact results prevents any conclusion about a preferable disposable cup material. The disposable PS cup life cycles do thus not perform better, but also not worse than the disposable biocup life cycles for PLA and biopaper (i.e. paper lined with bioplastic).

The large spread in our results were already presaged by the inconsistent and sometimes even conflicting results of the ten LCA studies in Van der Harst and Potting (2013). Each of these ten LCA studies always differed on more than one inventory data set and/or modeling choice with each of the other LCA studies. The influence and therewith importance of potential individual sources for spread in LCA results were therefore impossible to trace. Applying multiple inventory data sets for all major life cycle processes and multiple crediting principles for recycling allowed systematic quantification of their influence on the impact results for each of the ten cup life cycles. That is a major achievement of applying multiple inventory data sets and crediting principles for recycling, which in this LCA study generated a wealth of additional valuable scientific insights that are summarized here:

- The large spread in impact results, i.e. from applying multiple inventory data sets and crediting principles for recycling, hampers drawing decisive conclusions about the preferred disposable cup material. Such outcome in based on more robust impact results, however, than for those from LCAs based on single inventory data sets per life cycle process
- Despite their large spread, impact results consistently point to the same dominant processes in the life cycles for each disposable cup material. These dominant processes turned out to be the same as identified in the contribution analysis (i.e. none of the used inventory data sets made a given process into a minor contributor in the impact results for the cup life cycles)
- Particularly cradle to PLA production dominated the impact results for the four PLA cup life cycles. It should also be noted that PLA production was based on ‘cold’ PLA, since thermo-resistant PLA is not yet commercially produced
- The crediting of recycled material also considerably influenced the impact results for the recycling life cycles across all three disposable cup materials
- Across disposable cup materials, spread in impact results for energy related impact categories tend to be clearly smaller than in non-energy related impact categories (in the toxicity categories particularly)

- Average impact results for Abiotic Depletion and Global Warming are (obviously) better for the disposable biocups than PS cups, because PLA and paper are renewable materials and PS is not
- Production of all three disposable cup materials can environmentally improve, but this potential is probably largest for the relative new material PLA (presently produced from economically valuable sugar and starch instead of from lignocellulose in arable crops)
- Correlations between inventory data within one data set, e.g. between energy use and carbon dioxide emissions, are maintained by calculating spread on the basis of impact result. Mainstream LCA studies first calculate spread in inventory data, before performing impact assessment, which often violates existing correlations between inventory data within one data set

Table 2. Summary of the potentially misleading comparison of the average impact results for the disposable PS cup life cycle with incineration to the average impact results for the disposable PS cup life cycle with the recycling, and for the eight disposable biocup life cycles (i.e. PLA or biopaper in combination with one the four waste treatment processes; incineration = I, recycling = R, composting = C, anaerobic digestion = D). The results are potentially misleading because the spread around the average results is large and overlapping across all three materials in all impact categories. This is not reflected in the below indication whether average impact results for each of the other nine life cycles are higher than (>, darker grey shading), lower than (< , lighter grey shading) or similar as (1, similar grey shading) the average results of the disposable PS cups that is incinerated after use.

Impact category indicators	**PS**		**PLA**				**Biopaper**			
	I	**R**	**I**	**R**	**C**	**D**	**I**	**R**	**C**	**D**
Abiotic Depletion Potential (ADP)	1	<	<	<	<	<	<	<	<	<
Cumulative Energy Demand (CED)	1	<	>	<	>	>	>	<	>	>
Global Warming Potential (GDP	1	<	<	<	<	<	<	<	<	<
stratospheric Ozone Depletion Potential (ODP)	1	>	>	>	>	>	>	>	>	>
Acidification Potential (AP)	1	<	>	>	>	>	>	>	>	>
Eutrophication Potential (EP)	1	>	>	>	>	>	>	>	>	>
ground-level PhotoChemical Oxidation Potential (PCOP)	1	<	>	>	>	>	<	1	<	<
Human Toxicity Potential (HTP)	1	>	>	>	>	>	>	>	>	>
Terrestrial EcoToxicity Potenital (TETP)	1	>	>	>	>	>	>	>	>	>
Fresh-water Aquatic EcoToxicity Potential (FAETP)	1	>	>	>	>	>	>	>	>	>
Marine aquatic EcoToxicity Potential (MAETP)	1	>	>	>	>	>	>	>	>	>

3.2. Recycling slightly preferable over incinerating disposable cups
(Potting 2013; Van der Harst and Potting 2014; Van der Harst et al. 2014)

Table 3 shows the ranking of cup life cycles, within on disposable cup material, according to the average impact results for the used waste treatment process. Within one disposable cup material, the spread in impact results is identical from cradle to waste treatment entrance-gate, which allows focusing on the waste treatment processes only. There is also a large, but only partly overlapping spread in average impact results for the waste treatment processes within each cup material. Some cautious preferences are therefore possible to express on the basis of average results for the waste treatment process (i.e. not necessarily supported by the range in impact results for these processes.

Composting of the biocups is less good than the other three waste treatment processes as result of the absence of useful products derived from composting (e.g. both biocups do not contain nutrients). Composting therefore does not get credits, in contrast to the other three waste treatment processes, for the avoided production of products they replace. For the PLA cup, anaerobic digestion performs on average on almost all impact categories better than incineration for the PLA cup (i.e. avoided impact by energy production from biogas is larger than from incineration with energy recovery). There is no similar trend for the biopaper cup. The average impact results suggest a slight preference of recycling over incineration for the PLA cup and biopaper cup, i.e. the avoided impact of recycling is larger than for incineration, but not for the PS cups which average impact results are better in five and worse in six impact categories for recycling as compared to incineration. The comparison of recycling and incineration, however, revealed an interesting drawback of crediting processes with avoided production caused by their co-products.

The cup life cycles with incineration as waste treatment process got relative large credits for avoided electricity production from energy recovery by incineration, and these credits for incineration became more dominant when inventory data for improved disposable cup material production were used (i.e. credits for recycling became less). This may mistakenly suggests incineration to become preferable when disposable cup material production improves, whereas the appropriate policy recommendation would be to improve the relatively “dirty” Dutch electricity production. Dutch policies aim to improve the environmental performance of Dutch energy supply, amongst others by increasing the share of renewables (Government of the Netherlands 2014). Compared to other countries, Dutch electricity production uses little renewable sources, and predominantly relies on fossil fuels (CBS 2012; European Commission 2012; Eurostat 2012). A sensitivity analysis with hydro-dominated Norwegian instead of Dutch electricity production showed better impact results for recycling than for incineration of disposable PS cups in most impact categories.

The relatively large credits for avoided Dutch electricity production affected the comparison of cup life cycles with incineration and recycling as waste treatment process for all three disposable cup materials, but this particularly shows for the disposable PS cup in Table 3. Against this background, there is a slight preference for recycling over incinerating disposable cups. Other reflections worth to be mentioned here (Potting 2013):

- Pilot experiments suggest that efficiency and contamination of separate collection of disposable cups depends on the way of collecting (e.g. in stacks or loose in bins or containers). Contamination of the collected disposable cups was roughly 40%, 20% of beverage-remainders, and 20% others (e.g. plastic stirrer, plastic bread bags, metal staples, clock houses etc.)
- PLA is not compostable in a back-garden compost-heap, but disposable PLA and biopaper cups both composts well in (semi-) industrial compost facilities as a composting experiments showed (‘cold’ instead of thermo-resistant disposable cups were used in these experiments). Incomplete composted PLA cups remains visible as white traces, however, which makes the compost unfit for commercial sales
- Disposable biocups are in practice not composted in Dutch commercial composting facilities as biodegradable and non-degradable cups are difficult to distinguish, and also because incomplete composting of PLA cups leave white traces in the compost

Table 3. Ranking according to average impact results of waste treatment processes within disposable cup materials. Lowest impact results are indicated by 1, highest impact results are indicated by 4 (incineration = I, recycling = R, composting = C, anaerobic digestion = D).

Impact categories	PS		PLA				Biopaper			
	I	R	I	R	C	D	I	R	C	D
Abiotic Depletion Potential (ADP)	2	1	3	1	4	1	1	3	4	2
Cumulative Energy Demand (CED)	2	1	3	1	4	2	2	1	4	2
Global Warming Potential (GDP	2	1	3	1	4	2	1	3	4	1
stratospheric Ozone Depletion Potential (ODP)	1	2	1	3	4	2	1	2	4	3
Acidification Potential (AP)	2	1	3	1	4	2	2	1	4	3
Eutrophication Potential (EP)	1	2	3	1	4	2	3	1	4	2
ground-level PhotoChemical Oxidation Potential (PCOP)	2	1	3	1	4	2	2	1	4	3
Human Toxicity Potential (HTP)	1	2	3	2	4	1	3	1	4	2
Terrestrial EcoToxicity Potenital (TETP)	1	2	2	1	4	2	1	2	4	3
Fresh-water Aquatic EcoToxicity Potential (FAEP)	1	2	3	1	4	2	3	2	4	1
Marine aquatic EcoToxicity Potential (MAEP)	1	2	3	2	4	1	3	2	3	1

3.3. Dishwashing not convincing better than handwashing for reusable cups (Potting 2013)

Figure 1 shows the impact results of the screening LCAs of the reusable cup life cycles. The impact results are expressed as ratio with the average impact results for the disposable PS cup life cycle with incineration (for which the number of hot beverage consumptions is kept at a one). The reusable cup life cycle with dishwashing performs slightly better than the reusable cup life cycle with handwashing. The impact results for handwashing are strongly influenced, however, by the user-dependent amount of hot water, soap and paper towels applied in the screening LCA. These amounts were set on reasonable worst case amounts, but may in practice be considerably higher as well as lower. It is thus difficult to express a preference for either dishwashing or handwashing of reusable cups on the basis of these results.

As also can be seen from Figure 1, the impact results for both reusable cups roughly halves with two hot beverage consumptions before washing. The environmental gain declines with every next consumption before washing, however, and more than two or three consumptions does hardly lead anymore to a decrease of impact results. The disposable cups can of course also used for more than one consumption, and then a similar decline is at stake with every next consumption before disposing.

3.4. Reusable cups not better or worse than disposable PS (Potting 2013)

The impact results for the reusable cup life cycles from the screening LCA have been compared with the average impact results for the disposable PS cup life cycle with incineration from the in-depth LCA (for which the number of hot beverage consumptions is kept at one time). Similarly as reusable cups, disposable cups can also be used for more consumptions before disposing them. A fair comparison therefore should be based on using reusable and disposable cups for the same number of consumptions. This number of consumptions can be any, as long as it is similar between reusable and disposable cups. This comparison put the number on one consumption, however, since Figure 1 expresses the impact result for the reusable cups as ratio with the impact results for one consumption. Based one consumption for both reusable and disposable cup, the impact results for the reusable cup life cycle with dishwashing are better in some and worse in other impact categories, whereas the impact results for the reusable cup life cycle with handwashing are worse in all impact categories than the impact results for the disposable PS cup life with incineration. As already mentioned in Section 3.3, however, the impact results for the reusable cup life cycle with handwashing are strongly influenced by the user-dependent amounts of hot water, soap and paper towels applied in the screening LCA (which represent reasonable worst case amounts).

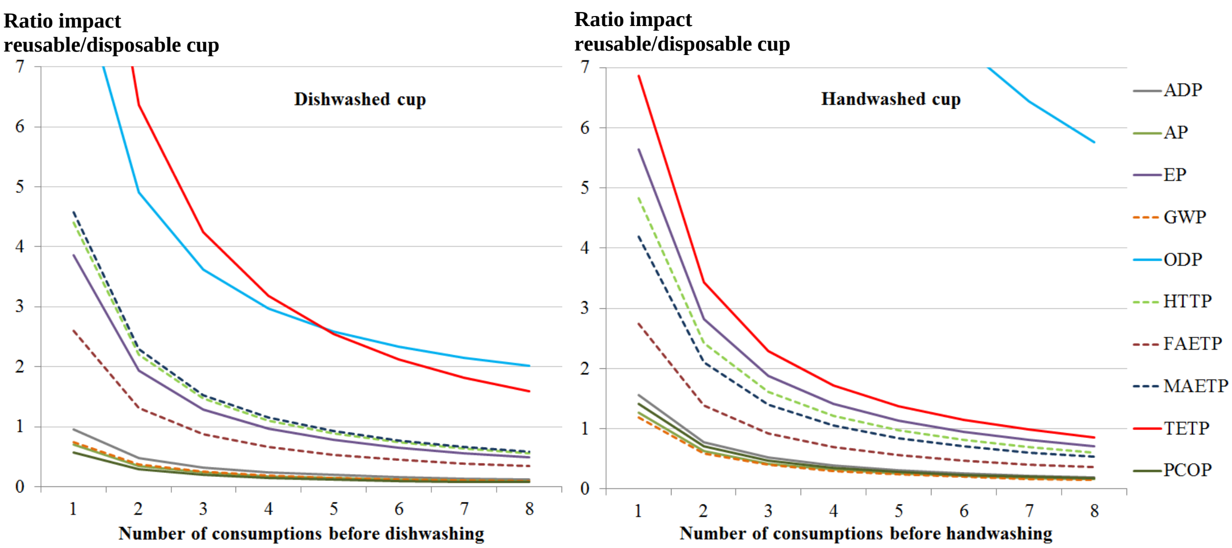

Figure 1. Impact results for the reusable cup life cycles, with dishwashing on the left and with handwashing on the right, expressed as ratio of the average impact results for the disposable PS cup life cycle with incineration as waste treatment (for which the number of hot beverage consumptions is kept one time)

3.5. Facility arrangements improving the environmental performance of all cups

While the overall comparison does not allow any preference for one of the three disposable cup materials, and neither for disposable versus reusable cups, Figure 1 does indicate for the reusable cups a considerable environmental gain from a second and possibly third hot beverage consumption with the reusable cups before washing it. This environmental gain obviously also exist for increasing the number of consumptions of the disposable cup before throwing it away. A second or third consumption is roughly the number of hot beverages that a consumer takes during one morning or one afternoon. Since pathogens probably not multiply so fast, and consumers usually do not share cups, there seems no real public health issue here. Facility arrangements can encourage a second or third serving with the same cup by financial incentives (e.g. paying for a new disposable cup), only putting on the dishwasher around noon and after working time, and/or consumer awareness activities. Consumer awareness activities should also point to the fact that more than two to three servings with the same cup hardly add environmental gain.

4. Conclusion

The overall results do not allow any preference for one of the three disposable cup materials (large and overlapping spread in impact results), and neither for disposable versus reusable cups (impact results for the latter too uncertain and too close to those for the disposable cups). All cups can be used more than once before getting rid of a disposable cup or washing a reusable cup. This gives a considerable environmental gain for the second and third hot beverage consumption with the reusable as well as disposable cups. Facility arrangements can encourage a second or third serving with the same cup by financial incentives (e.g. paying for a new disposable cup), only putting on the dishwasher around noon and after working time, and/or consumer awareness activities. Consumer awareness activities should also point to the fact that more than two to three servings with the same cup hardly add environmental gain.

It was not possible to indicate a preference for one of the three disposable cups, but comparison of waste treatment processes for each cup material gave some basis to express some preferences on the basis of average impact results. Composting is the least preferred waste treatment for both biocups. Anaerobic digestion performs better than incineration for the disposable PLA cup in most impact categories, though this trend does not apply for the biopaper cup. The average impact results suggest a slight preference of recycling over incineration for the PLA cup and biopaper cup, i.e. the impact of recycling is smaller than for incineration, but not for the PS cups which average impact results are better in five and worse in six impact categories for recycling as compared to incineration. The comparison of recycling and incineration, however, is biased by the relative large credits for avoided “dirty” Dutch electricity production. Against this background, there is a slight preference for recycling for all three disposable cup materials.

The in-depth LCA study comparing the disposable cups deliberately applied multiple inventory data sets for the processes contributing most to the impact results, as well as multiple crediting principles for recycling. This led to a large spread in impact results, though for energy related impact categories smaller than for the others. The large spread in impact results may be less easy to interpret, but they represent more robust results.

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Introduction of grass-clover crops as biogas feedstock in cereal-dominated crop rotations. Part I: Effects on soil organic carbon and food production

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ABSTRACT

Changes of soil organic carbon (SOC) content can have a substantial effect on greenhouse gas emissions, but are rarely included in crop production LCAs. SOC content strongly influences soil fertility and therefore crop yields, but is declining in many European soils. The present study investigated if integration of 1-2 years of grass-clover crops in a cereal-dominated crop rotation can increase the SOC pool and how this would impact food production. Results show that when grass-clover crops are integrated, the potential SOC content at steady state will be 41 to 52% higher than in the conventional cereal-dominated crop rotation. The net increase of wheat yields based on SOC improvements indicate that for a crop rotation with one year of grass-clover crops, the initial loss of food production can be counterbalanced due to the impact on fertility of the SOC increase.

Keywords: soil organic carbon, grass-clover crops, food crop production, crop yield, soil carbon model

1. Introduction

Energy crop production is often claimed to negatively impact food and feed production, i.e. by competition for arable land and by reducing soil quality. On the other hand, current intensively managed agricultural food and feed production systems are often not sustainable: Loss of soil organic carbon (SOC), erosion and compaction are the main processes threatening soil fertility throughout the EU (EC 2002; Soilservice 2012). Intensively cultivated clay soils have in Swedish studies been shown to give up to 20% decreasing food crop harvest yields due to soil compaction and reduced soil organic matter content (Arvidsson and Håkansson 1991).

The build-up or degradation of SOC is a process with dual and important impact on sustainability in crop production. Firstly, SOC content strongly influences soil fertility and crop yields, but due to e.g. unsustainable management practices and crop rotations it is declining in many European soils (Soilservice 2012). Secondly, agricultural soils can act either as carbon sinks, or, if SOC is declining, as contributors to greenhouse gas emissions. Changes in SOC can contribute substantially to greenhouse gas emissions of crop production, but are rarely included in LCA (Brandão et al. 2011).

SOC content is positively influenced by addition of organic materials such as intermediate crops, crop residues and manure or other organic fertilizers. The main negative impacts on SOC are the outflow caused by mineralization processes of the soil organic matter, the degree of which is related to e.g. climatic conditions and the severity of soil management practices. By changing crops in the crop rotation SOC can be influenced actively. However, addition of organic material is often small in comparison to the amount of carbon already stored in the soil. Therefore long-term changes of SOC are often quantified using models that are calibrated with long-term experimental data.

The present study investigated if integration of 1-2 years of grass-clover crops in a cereal-dominated food crop rotation on intensively managed heavy clay soils can increase the SOC pool. The work is presented in two parts, where the present paper, Part I, shows the impacts on the SOC pool and discusses the potential impact on fertility and crop yields. Part II shows the LCA and effects on greenhouse gas emissions of grass-clover crop introduction to the crop rotation (Björnsson and Prade 2014). In areas with little animal husbandry there is no or a very limited market for grass-clover crops as animal feed and limited availability of organic fertilizers such as manure. Therefore, the approach in this study was to integrate grass-clover crops in the crop rotation for use as biogas feedstock. The biogas plant then takes on the role of the absent ruminants, creates a market for the grass-clover as energy crop, and produces biogas and a digestate (the liquid residue of the biogas production), which is used as biofertilizer.

The study was performed as a farm based case study, where the case farm is representative for a large agricultural region in which the low input of carbon in the cereal based crop rotation has been identified as problematic with regards to the high clay content in the soil. The hypotheses of this study were that (a) an increased SOC content can be obtained by introduction of e.g. grass-clover crops as well as by fertilization with biogas digestate; and (b) that the increased SOC content may increase food crop yields enough to compensate the initial food production losses when the crop rotation is changed.

2. Materials and methods

2.1. The case farm

The farm based case study investigates the effects of different crop rotation and biomass utilization scenarios. As a basis for this case study, a farm with mainly clay rich soils in North West Scania (56°6'N 12°58'E) was chosen. This farm, Wrams Gunnarstorp, is located close to the site of long-term SOC content field experiments in Ekebo, performed by the Swedish University of Agricultural Sciences (SLU) (Kirchmann et al. 1999).

This study concentrated on 650 ha of medium to heavy clay soils with soil clay content up to 65%. The soils are rather cold, and crop establishment is often carried out very shortly in the autumn after harvest of the previous crop. Crop establishment is rather slow, and the risk for the plants to be too big for overwintering is little. This leaves no opening for the introduction of after-sown intermediate crops. In this study, introduction of clover-grass crops extending the crop rotation was chosen as a measure to increase SOC content.

The latest analysis of soil carbon content expressed as humus on the Wrams Gunnarstorp farm dated from 1984. That year, the soils had an average SOC content of ~2% and consisted of very heavy clay soils, heavy clay soils and medium clay soils. The fairly high content of humus in the Wrams Gunnarstorp soils at this time was probably a result of the consequent use of cow manure as biofertilizer until 1960.

2.2. Crop Rotation

On the major part of the farm, a 4-year crop rotation typical for the region is used, Table 1. This present crop-rotation was used in the reference scenario. In the investigated alternative scenarios, grass-clover crops were introduced one year in a five year crop rotation (scenario GC1) and two years in a six-year crop rotation (scenario GC2).

Table 1. Typical sowing and harvest dates on the Wrams Gunnarstorp farm for the reference crop rotation (years 1-4) and extension with grass-clover crops.

Year	Crop	Sowing date	Harvest date
1	Winter oilseed rape	1-10 August	20 July
2	Winter wheat	1-20 September	10 August
3	Winter wheat	1-20 September	10 August
4	Oats	1-20 April	20 August
Scenario GC1 years 1-4 plus			
5	Grass-clover crops	1-20 April[a]	Cut 1: 1 June; cut 2: 1 August
Scenario GC2 years 1-4 plus			
5	Grass-clover crops	1-20 April[a]	Cut 1: 20 June; cut 2´: 20 August
6	Grass-clover crops	-	Cut 1: 1 June; cut 2: 1 August

[a] Undersown in oats in the previous year

2.3. Crop production

Average crop yields were estimated based on annual measurements on the Wrams Gunnarstorp farm. For cereal grains and oilseed rape average production yields were calculated, Table 2. Yields for grass-clover crops were estimated from hand-harvested samples and corresponding machinery field losses (20%) within an ongoing research project at SLU, evaluating grass-clover crop yields on the Wrams Gunnarstorp farm (funded by Stiftelsen Lantbruksforskning, SLF). A grass-clover crop system with two harvests per year was assumed.

Grass-clover crops were assumed to be undersown with the preceding crop, oats. After the oats have been harvested, the grass-clover crops grow up. In the year of establishment, grass-clover crops are assumed to be harvested once in the late autumn. Oats are assumed to be harvested 20th of August and grass-clover crops harvested 30th of September are assumed to result in a biomass yield of 1.5 t ha^{-1} of DM. Winter wheat is assumed to be harvested 10th of August. Breaking of the grass-clover crop is assumed to be carried out 1st of August in order to allow establishment of winter oilseed rape (WOSR). In this year, the grass-clover crop is assumed to yield 9.0 t ha^{-1} instead of 12 t ha^{-1} DM in a full production year. A full production year is only possible the first year of a two-year grass-clover crop.

Details on inputs in crop production (machinery, materials, diesel, fertilizer) are based on typical cultivation input and fertilization levels at the model farm and are presented in Part II of the study, where greenhouse gas emissions for the different scenarios are calculated.

2.4. Amounts of crop residues

Crop yields play a central role in this study, since many analysis parameters are directly or indirectly connected to biomass and/or grain yields. Higher crop yields often result in larger amounts of crop residues, e.g. straw, stubble, roots and extra root biomass, which will impact SOC input. Most models for calculation of crop residues assume a linear connection between harvestable biomass (i.e. grains, seeds, beets, above-ground biomass) and remaining residues above- and belowground in the form of fixed mass ratios for the different plant parts. This is the case for the calculation model for amounts of crop residues suggested by IPCC (2006).

Swedish long term field experiments support models that result in high biomass respective carbon inputs from root and extra root material, higher than what is suggested in the IPCC model (Björnsson et al. 2013). This is especially valid for grass-clover crops, where a large number of plant species of grasses and legumes can be mixed in endless combinations. While grasses contribute much harvestable biomass, legumes contribute nitrogen fixation and root biomass. Another aspect of grass-clover crops is the time factor. High production systems may utilize grass-clover crop blends for 1-3 years, while more long-term or permanent grass-clover crop systems exist as well.

Swedish studies fitting long-term soil carbon measurements to a soil carbon model suggest a constant amount root biomass, 6 t ha^{-1} of DM (Bertilsson 2006). However, in this study, a proportional root biomass development was assumed in the base case, limited with a ceiling value of 6 t ha^{-1} of DM. Another issue is that straw yields suggested in the IPCC calculation model are unrealistically high compared to actual straw yields in cereal cultivation in under Nordic conditions (Nilsson and Bernesson 2009).

In the base case calculations of SOC, model parameters from regional studies (called Nordic) are used for the calculation of amounts of crop residues as described above (Table 2) (Björnsson et al. 2013). For comparison the calculations were repeated with the IPCC methodology for crop residue calculation (Table 2).

Table 2. Dry matter (DM) yield data of harvested crop parts (grains, seeds, grass-clover cuttings) and coefficients used in the systems analysis and for calculation of crop residues and SOC. (1 t = 10^6 g; 1 ha=10.000 m^2).

Crop	DM yield	Nordic		IPCC (IPCC 2006)			Humification coefficients[c]	
	[t ha^{-1}]	Slope	B/A ratio[a]	Slope	Intercept[b]	B/A ratio[a]	Above-ground	Below-ground
Winter oilseed rape	2.5	0.92	0.20	1.09	0.88	0.22	0.15	0.35
Winter wheat	6.5	0.57	0.33	1.61	0.40	0.23	0.15	0.35
Oats	4.0	0.50	0.47	0.91	0.89	0.25	0.15	0.35
Grass-clover, 0 year after oats	1.5	0.25	0.58[d]	0.30	0.00	0.00[d]	0.12	0.35
Grass-clover, 1st of one year	9.0	0.25	0.88	0.30	0.00	0.80	0.12	0.35
Grass-clover, 1st of two years	12.0	0.25	0.26[d]	0.30	0.00	0.00[d]	0.12	0.35
Grass-clover, 2nd of two years	9.0	0.25	0.88	0.30	0.00	0.80	0.12	0.35

[a] Belowground residues/aboveground biomass ratio: aboveground includes stubble and harvested biomass; belowground in Nordic includes extra-root material.

[b] DM in [t ha^{-1}]

[c] (Kätterer et al. 2011); Digestate: 0.41

[d] Only extra-root biomass is accounted for.

Another carbon input with relevance for the SOC calculations in the GC1 and GC2 scenarios is the added biofertilizer, the digestate, which is the residue from the harvested above ground part of the grass-clover crops after biogas production. The details of biogas production are presented in Björnsson and Prade (2014). The data used in the SOC calculation in the present part of the study are presented in Table 3.

Table 3. Harvested grass-clover crop biomass for use as biogas feedstock and resulting digestate for the grass-clover scenarios.

Parameter	Unit	Scenario GC1		Scenario GC2	
		Biogas feedstock[a]	Digestate[b]	Biogas feedstock[a]	Digestate[b]
Dry matter	[kg ha^{-1} a^{-1}]	1995	867	3563	1543
Carbon	[kg ha^{-1} a^{-1}]	904	430	1614	766

[a] Feedstock indicates the average amount of grass-clover removed in the crop rotation after field drying to 35% DM and ensiling.
[b] Digestate indicates what is returned to the field as biofertilizer in average for the whole crop rotation after biogas production and losses during digestate storage.

2.5. SOC model

In order to calculate changes in the SOC content as influenced by the choice of crop rotation, Introductory Soil Carbon Balance Model (ICBM) was used (Andrén and Kätterer 1997; Kätterer and Andrén 2001). The model was applied to calculate the SOC content according to carbon inputs and mineralization rates. The model was modified to account for different input types with specific humification factors (Figure 1).

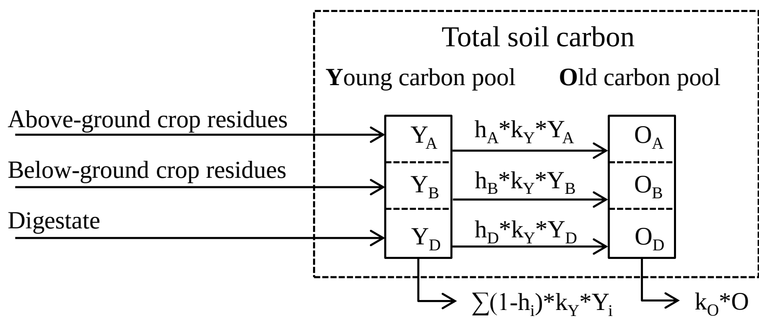

Figure 1. Introductory Soil Carbon Balance Model (ICBM)

All carbon from aboveground residues (A), belowground residues (B) and digestate additions (D) enters the young carbon pool (Y). From this pool, easily degradable carbon is released as CO_2 by mineralization according to a degradation function, while only a material-specific fraction is humified, i.e. stored in the humus part of soil carbon, or, in this model, the old carbon pool. Y has an outflow of carbon with a relatively high reaction coefficient of k_Y=0.8, i.e. within one year 1-exp(-0.8)=55% of the carbon leaves the young carbon pool again (Andrén and Kätterer 1997). The output from the old carbon pool follows a much lower reaction coefficient (k_O) than the young carbon. Carbon content of crop dry matter was assumed to be 45% (Kätterer et al. 2011). A starting value of 2% SOC content was assumed. In the base case, annual SOC content changes were calculated as average values over a time span of 40 years.

2.6. Model calibration

In order to adapt the ICBM to Nordic conditions, the model was calibrated against data derived from the long-term soil carbon field experiment in Ekebo, Sweden (Kirchmann et al. 1999). The Ekebo soil carbon field experiment includes two different crop rotations. One was designed as a crop rotation for an animal production farm, with all cereal straw and sugar beet tops removed. The other crop rotation was designed for a pure plant production farm, with all straw and sugar beet tops left in the field. For each rotation 16 different fertilization regimes (all combinations of 4 nitrogen and 4 phosphorous/potassium fertilization levels) were tested. The experiment started 1957 and is ongoing with regular soil carbon content analyses.

Calibration was done using the reaction coefficient of the old carbon pool (k_O) as a variable to fit model soil carbon predictions to the measured soil carbon data. This was done using crop residue data as computed by (a) the IPCC calculation and (b) by the Nordic calculation for comparison. The prediction power of the model was computed by maximizing the coefficient of determination (R^2) of the measured and predicted data.

2.7. Yield impact

Food crop grain and seed yields have been shown to increase corresponding to the SOC content (Lal 2004; Soilservice 2012). Yield changes due to SOC increases have been calculated for wheat grain yields assuming a DM yield increase of around 0.4-0.8 t ha^{-1} $\%_{SOC}^{-1}$ (Lal 2004). For the SOC content of the case farm soils, each percent of SOC corresponds to approx. 20 t ha^{-1} carbon.

2.8. Sensitivity analysis

In the base case, calculations follow the Nordic methodology, while in the sensitivity analysis IPCC methodology was applied (Table 4).

Table 4. Parameters use in the base and the alternative case.

Parameter	Nordic – base case	IPCC – alternative case
Aboveground crop residues	Only slope	Slope and intercept
Belowground crop residues	Limited to 6 t ha^{-1} of DM; extra root residues correspond to 65% of root residues[a]	Unlimited
Time span for calculation of average annual carbon changes	40 years	20 years

[a] (Bolinder et al. 2007)

3. Results

3.1. SOC changes

The soil carbon content develops positively for all scenarios, but with largely different long-term results, Figure 2. The reference scenario has the conventional cereal based 4-year crop rotation, where all of the straw is left in the field. In this scenario, the steady state of SOC content was reached at 2.9% after approx. 145 years. In the GC-scenarios, grass-clover is introduced in the crop rotation, and the amount of digestate produced from grass-clover digestion is large enough to cover substantial shares of the fertilizer need at the farm, i.e. 15 and 24% of nitrogen, 23 and 34% of phosphorous and 42 and 50% of potassium for scenarios GC1 and GC2, respectively (Björnsson and Prade 2014). The GC-scenarios contribute to a much higher increase in SOC content with steady states at 4.1 and 4.4% after approx. 125 years for scenarios GC1 and GC2, respectively.

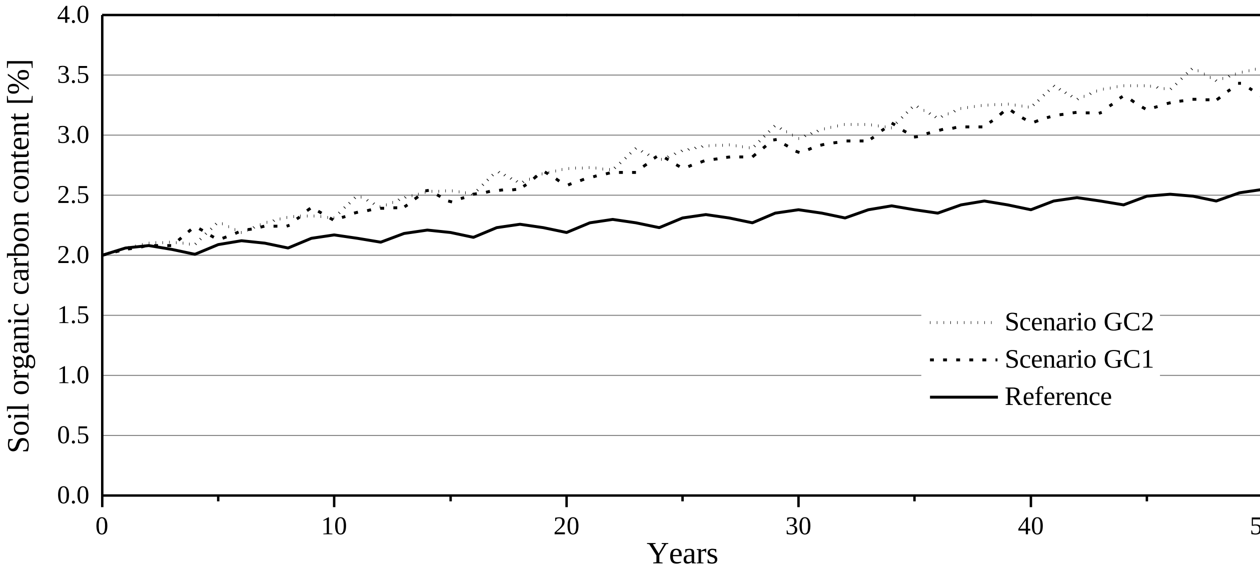

Figure 2. Change in soil carbon content in the investigated scenarios over a 50-year period according to base case calculations.

Figure 3 shows the average annual SOC changes for the crops and the crop rotations studied calculated both with the Nordic and the IPCC methodology. For oilseed rape, oats and grass clover crops, the Nordic methodology results in lower annual SOC additions than the corresponding IPCC methodology, but with relatively small differences. For wheat, using the IPCC methodology results in 168% higher crop residue inputs.

These differences are reflected in the average annual SOC changes calculated for the complete crop rotations. Using the IPCC methodology resulting annual SOC changes are 170%, 47% and 18% higher for the reference scenario, GC1 and GC2 respectively than using the Nordic methodology.

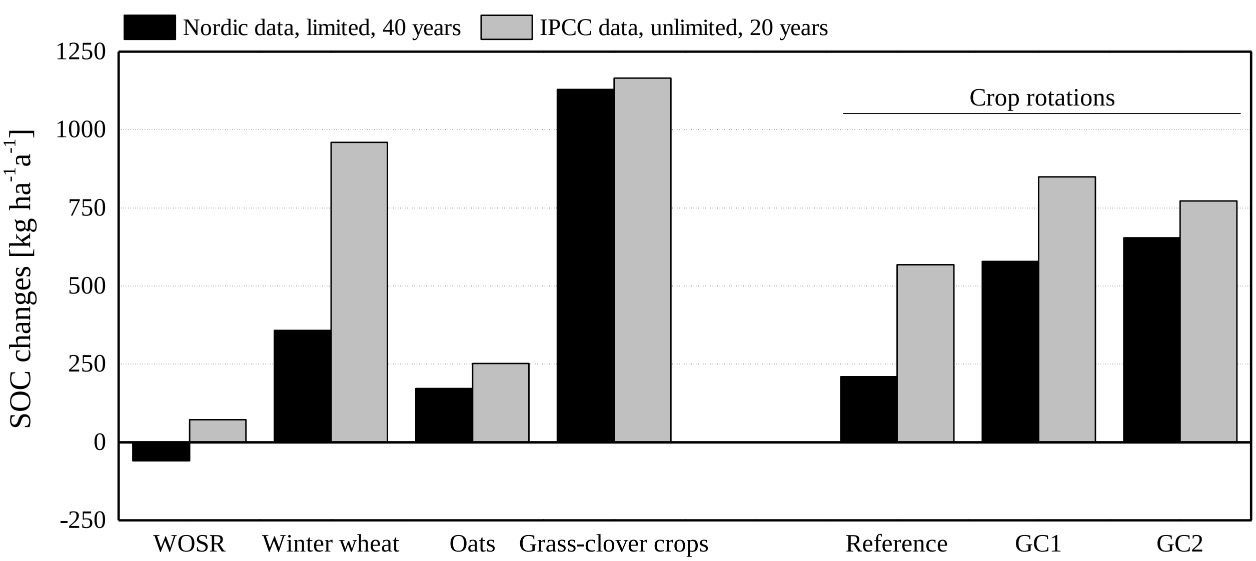

Figure 3. Annual SOC changes of crops and crop rotations according to Nordic and IPCC methodology. For crops, continuous cultivation was assumed. Data for crop rotations represent average annual SOC changes from crop residues and digestate application. 'Limited' and 'Unlimited' refer to root SOC contribution for grass-clover crops. Years given refer to the time span used for the calculation of the average annual SOC changes. WOSR=winter oilseed rape.

The SOC content after 50 years and at steady state (SS) calculated both with Nordic and IPCC methodology are shown in Figure 4. With the rapid and large SOC increase in the GC1 and GC2 scenarios (Figure 2), SOC after

50 years will be more than 0.6-0.8% higher (GC1) to 0.5-1.0% higher (GC2) than in the reference scenario.

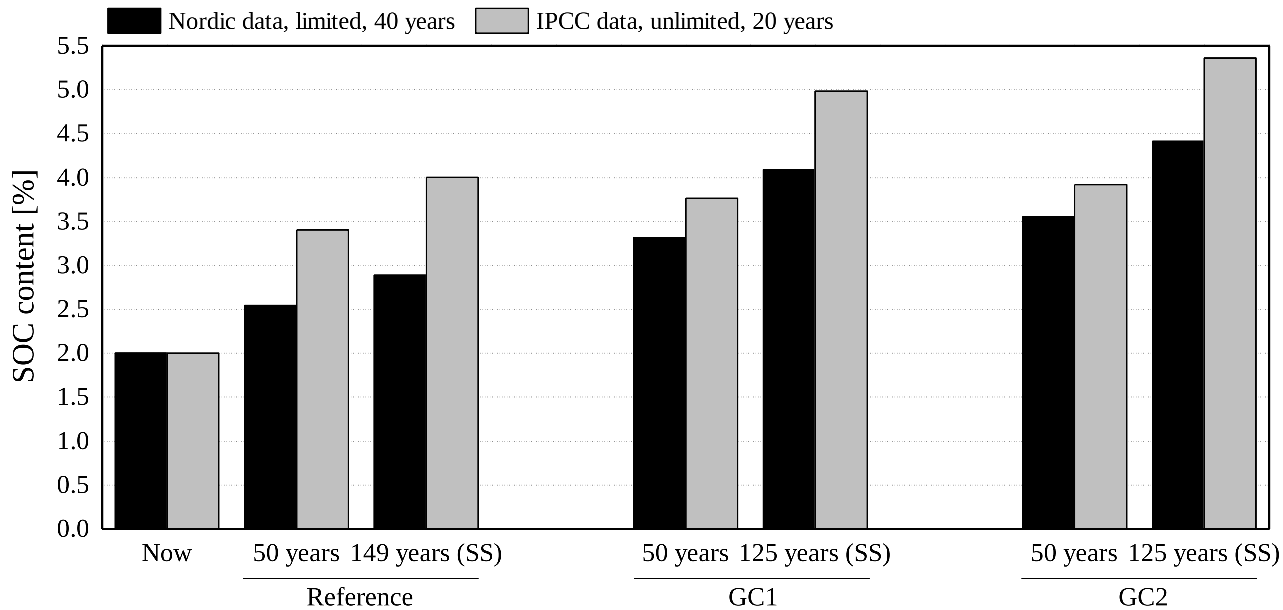

Figure 4. Predicted SOC content at the case farm after 50 years and at steady state (SS) for to the different scenarios according to the Nordic and IPCC calculation methodologies.

3.2. Yield changes

The SOC values after 50 year (Figure 4) would mean 5-10% and 6-12% net increase in yield of wheat grain for scenarios GC1 and GC2, respectively compared to yields in the reference scenario (Figure 5). At steady state, the decrease in food crop production of 20 % - which is the consequence of introducing one year of grass-clover crops in the crop rotation (scenario GC1) - could be offset to a major extent by the yield increase. In scenario GC2, the yield increase at steady state would correspond to around half of the initial food production losses for Nordic and IPCC approach.

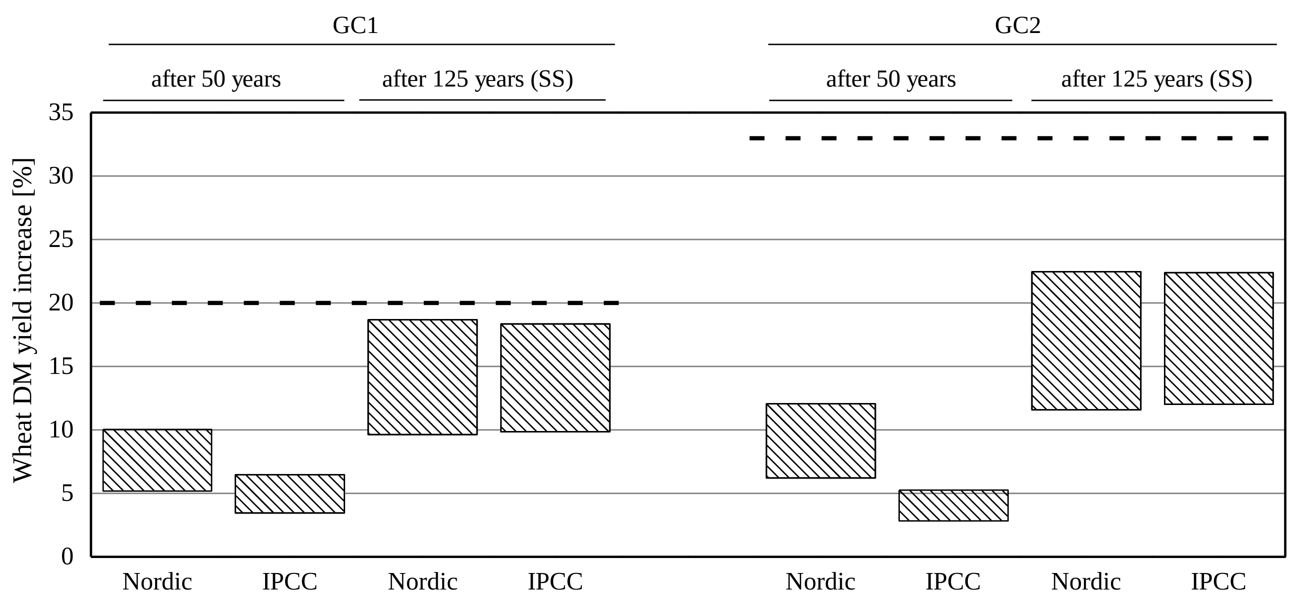

Figure 5. Predicted wheat yield increase relative to the reference scenario at the case farm after 50 years and at steady state (SS). Dashed lines correspond to the initial food production losses caused by the crop rotation changes in the scenarios.

4. Discussion

4.1. SOC changes

SOC content is a parameter that connects soil fertility, crop yields and greenhouse gas issues and therefore is a crucial factor for the environmental performance of a cropping system. Processes leading to changes in carbon soil content are often slow; therefore the time perspective becomes important for life-cycle assessment of agricultural production systems.

This study has shown that extension of a cereal-dominated crop rotation with one or two years of grass-clover crops can improve SOC content in a heavy clay soil substantially, given that the change is a medium to long-term commitment. In fact, the case farm currently operates a second crop rotation on a smaller fraction of the farm, including three years of meadow fescue for seed production in order to decrease problems associated with heavy clay soils and to improve soil fertility. However, the area on which this improved crop rotation is used is limited by the marketing possibilities for meadow fescue seeds. A biogas plant in a region characterized by crop production and lack of organic fertilizers could help create a potential market for grass-clover crops and in return deliver digestate, the liquid effluent from the biogas process, for utilization as organic fertilizer. The results further indicate that improvements of SOC content by including one year of grass-clover crops can potentially offset the initial food production losses caused by the crop rotation change.

Soil carbon content affects many cultivation factors such as nutrient availability, water retention capacity, soil density, soil temperature etc. An increase of soil carbon content may lead to, but is no guaranty for, increased soil productivity, i.e. increased crop yields. But also decreased requirements for fertilization can be a positive result of soil carbon increase. Other, more short-term effects may contribute positively to food production in the system, e.g. the pre-crop effect of grass-clover crops.

Another strategy for improving SOC content would be to implement grass-clover crops in order to boost SOC content to the steady state of the reference scenario in only 29 and 26 years for scenarios GC1 and GC2, respectively, e.g. if return to the crop rotation in the reference scenario is desired.

4.2. Yield increases

The estimates of food crop yield increases in the present study are rather conservative, since calculations neglect two effects. First, in the short term, soil structure improvements by grass-clover crop cultivation could help decrease yield losses due to soil compaction and standing water. Secondly, these short-term yield increases and SOC-related yield increases have a positive impact on crop residue input. Therefore, these calculations underestimate potential yield increases. Therefore, the next step for a follow-up project is to implement yield impacts, accumulated food production losses and impacts on fertilization requirements in the SOC model.

4.3. Calculation methodology

While changes in a crop rotation have a small short-term effect, the long-term effects may lead to significant changes in soil fertility, crop yields and greenhouse gas emissions of a cultivation system. It is therefore important to investigate potential long-term effects of carbon input on the soil carbon content. IPCC suggests a period of 20 years to be used for this purpose. In theory, changing from one established cropping system to another with a change in the average carbon input may result in a change of soil carbon content. How quickly the change will take place and the time required to reach a new steady state depends on many factors. However, the soil carbon content will change along an asymptotic curve, with the highest absolute changes in the beginning. With time, these differences will become smaller and smaller until a new steady state is reached. Therefore, the shorter period for calculating the average annual carbon change as suggested by IPCC will result in higher changes compared to a longer calculation period and may therefore potentially overestimate the effect of crop residue additions.

Another issue with the IPCC calculation methodology is the overestimation of wheat crop residues, at least compared to Nordic conditions. The straw yields suggested in the IPCC calculation model are unrealistically high compared to actual straw yields in cereal cultivation in Sweden (Nilsson and Bernesson 2009). IPCC data gives thus higher SOC accumulation for all the investigated scenarios, but with larger impact on the reference scenario with 75% cereals in the crop rotation.

5. Conclusions

The investigated change from an agricultural food production system only producing food crops to a system with integrated production of food and energy crops was shown to be a potentially important tool to improve SOC content considerably. Securing future food production will require sustained or even increased SOC content. While extending food crop rotations with grass-clover crops results in a direct decline in food production, the long-term effect of SOC increase may increase food yields, both due to improved soil fertility and structure.

Extending cereal-crop dominated crop rotations with grass-clover crops could help integrate biofuel and food production. Plant nutrients in the biogas digestate can be recycled, including SOC building carbon. This diversification of crop rotation is especially interesting in regions with no demand for cattle feed.

Finally, carbon sequestration by increasing SOC content and greenhouse gas mitigation by biogas production from grass-clover crop biomass may improve the carbon balance of food production considerably, which is further discussed in Part II of this study (Björnsson and Prade 2014). This study demonstrated the importance of SOC changes in agricultural production systems and such changes should be accounted for in future LCA studies.

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Environmental assessment of bioethanol from onshore grown *Ulva*

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ABSTRACT

Besides biofuels from microalgae, an emerging interest in using macroalgae as feedstock for biofuel production is observable. Macroalgae have the advantage that they are much easier to harvest than microalgae so that the problem of low feedstock concentration does not arise. The environmental performance of bioethanol from onshore grown green algae is assessed using literature data and initial laboratory scale data. The optimized system model allows for producing an environmentally efficient biofuel in comparison to fossil fuel and bioethanol from sugar cane. Handling the co-product by substitution instead of energy allocation significantly reduced the environmental impacts of the system and resulted in environmental bonuses in several impact categories. Thus, the management of the co-product in the LCA model (energy allocation vs. substitution) is a key step in the LCA, as it highly influences the impact assessment results.

Keywords: Bioethanol, macroalgae, *Ulva*, eco-design, allocation, substitution

1. Introduction

Algae (microalgae and macroalgae) have recently been identified as an attractive, alternative renewable source for biofuel production compared to biomass from food or cellulosic materials (John et al. 2011; Wei et al. 2013). Marine algae production does not compete with food production, as algae do not need fresh water or arable land, but may use land depending on the culture system selected. Algae can use CO_2 from industrial emissions as carbon source. In addition, the biomass yield per unit area is higher than that for terrestrial biomass (Gao and McKinley 1994). While the production of biofuels from microalgae is intensively studied since a couple of years, the interest of using macroalgae as feedstock for biofuel production just emerges. Macroalgae have the advantage that they are much easier to harvest than microalgae so that the problem of low feedstock concentration does not arise.

To our knowledge, only a few Life Cycle Assessment (LCA) studies have been carried out so far with respect to biofuel production from macroalgae. Pilicka et al. (2011) evaluated the environmental impact of biogas production from onshore cultivated macroalgae. Langlois et al. (2012) studied environmental effects of biogas production from offshore grown seaweed. Alvarado-Morales et al. (2013) conducted an LCA study of biofuels from offshore grown brown algae in Nordic conditions, focusing on biogas production, and both bioethanol and biogas production. Aitken et al. (2014) assessed environmental burdens of biofuel production from offshore grown macroalgae, focusing on different offshore cultivation methods and the production of biogas and biogas + bioethanol.

This study assesses the environmental performance of bioethanol from onshore cultivated green macroalgae (sea lettuce). The evaluation was based on literature data and initial laboratory scale data, as industrial scale facilities for bioethanol production from macroalgae do not exist. Limits of this approach were discussed with a focus on the co-product management in the LCA model.

2. Methods

2.1. Goal and scope, and functional unit

The goal of the study is to evaluate the environmental performance of bioethanol production from onshore grown macroalgae (*Ulva* sp.) using the Life Cycle Assessment (LCA) methodology. A reference scenario (Figure 1) is assessed to determine the main contributors to the environmental impact of the system. Based on this contribution analysis, an optimized system is proposed with eco-design improvements. The functional unit of the system is the production of 1 MJ by combustion in a passenger car in order to compare the environmental impact of the combustion of bioethanol from *Ulva* with that of other fuels. Environmental impacts are assessed at midpoint level using the ILCD 2011 impact assessment method (European Commission - Joint Research Centre - Institute for Environment and Sustainability 2011).

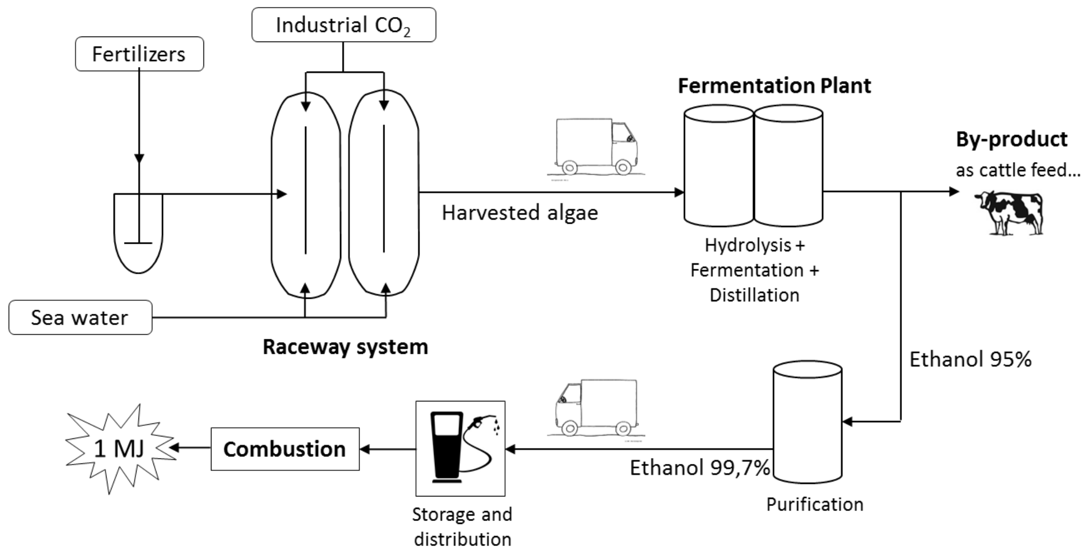

Figure 1. Schematic representation of the process chain for bioethanol production from onshore grown macroalgae.

2.2. Life cycle inventory

The life cycle inventory is based on literature data and initial laboratory scale data, as industrial scale facilities for bioethanol production from macroalgae do not exist.

2.2.1. Seaweed production

Algae (*Ulva* sp.) are cultivated in open raceways, and are stressed by nitrogen starvation to obtain high starch contents (up to 40% of the dry matter). In total, 8 raceways, each with a surface area of 500 m^2 and a water volume of 250 m^3, are used for seaweed production. Four raceways are used for growing *Ulva* at high nitrogen concentrations (high biomass yields), while the other four raceways are used for nitrogen starvation of *Ulva*. The excavated raceways are lined with an EPDM liner of 1 mm thickness and are mixed by paddle wheels. The productivity of *Ulva* is estimated to be 20 g dry weight (DW)/m^2/day with an initial algae density of 3 kg fresh weight (FW)/m^2. Assuming that the *Ulva* production site is located in Brittany, France, the *Ulva* production season has a length of seven months, ranging from April to the end of October. A season of 28 weeks is assumed, resulting in an *Ulva* production of 3.92 kg DW/m^2/season (39.2 t DW/ha/season). The seawater in the raceways is exchanged once a week with fresh seawater, filtered by a drum filter. Nutrients are supplied once a week to the first raceway system by dosing a modified f/2 culture medium (without vitamin and silicium solutions) at recommended quantities (Andersen 2005). The nutrient solutions are stored in three stainless steel tanks with an effective volume of 1 m^3 each. The tanks are mixed during the night (12 hours) to ensure homogenization of the nutrient solutions. Compressed and liquefied CO_2 is injected into the raceways through PVC pipes. Seaweed is harvested using perforated conveyor belts.

2.2.2. Bioethanol production

The harvested seaweed is transported over 60 km from its production site to the bioethanol production plant. Ethanol is produced by hydrolyzing and fermenting starch, followed by a distillation. The ethanol production process is based on ethanol production processes inventoried in the EcoInvent v2.2 database and described in detail in Jungbluth et al. (2007). Data for input of chemicals and other materials, energy consumption (electricity and heat), needed equipment, and emissions to the environment are determined as dry weight-based averages from EcoInvent inventories for bioethanol production from other starch-rich biomasses (potatoes, rye, and corn). The bioethanol yield of *Ulva* is estimated to be 0.13 g bioethanol/g DW *Ulva*.

2.2.3. Bioethanol purification, storage, distribution, and combustion

Purification, storage, distribution to service stations, and combustion of bioethanol in a passenger car are based on processes inventoried in the EcoInvent v2.2 database for bioethanol from other feedstocks.

2.2.4. Valorization of by-products and allocation

The residue of the fermentation step (named Distiller's dried grains with solubles (DDGS) by analogy) is assumed to be valorized as animal feedstock, as it is done in EcoInvent v2.2 for fermentation residues issued from other starch-rich biomasses (potatoes, corn, rye) (Jungbluth et al. 2007). The production of 1 kg ethanol from *Ulva* generates 5.74 kg (dry weight) of a DDGS equivalent. The co-product is handled by energy allocation as recommended by the European Commission for biofuels (European Commission 2009). A lower heating value (LHV) of 28.1MJ/kg is assumed for ethanol (Jungbluth et al. 2007). For the by-product, a LHV of 13.5 MJ/kg is estimated based on the LHVs of its components. This results in an environmental burden of 26.6% for the produced bioethanol and 73.4% for the by-product.

3. Results

The contribution analysis for producing 1 kg bioethanol with the reference system showed that the main contributors to the environmental impact of the system were the electricity consumption (French electricity mix: 78.1% nuclear, 10.8% hydroelectric, 4.4% coal, 3.2% natural gas, 1.5% other fossil fuels, 2.0% other), the infrastructure of the macroalgae production system (scale-up effect), and the origin of the nutrients for macroalgae production. Based on these results, an optimized system was defined, which had a modified algae production infrastructure (raceway dimensions and choice of materials) resulting in reduced electricity consumption and used fish farm wastewater as nutrient source for macroalgae.

The improved system model significantly reduced the environmental impacts of bioethanol production from onshore grown macroalgae (Figure 2). While the reference system had the highest environmental impacts on 12 of the 16 impact categories, the optimized system did not have the highest impact on any impact category. With the optimized system, bioethanol from macroalgae can be produced at lower environmental burdens than from sugar cane. The large impact of bioethanol from macroalgae on ionizing radiation resulted from the consumption of French electricity for macroalgae production. Using electricity from offshore wind power plants drastically reduced the impact on ionizing radiation to a level similar for petrol or bioethanol from sugar cane (data not shown). Macroalgae based biofuel combustion (optimized scenario) reduced greenhouse gas emissions by 57% and ozone depletion by 67% compared to fossil fuel combustion. In general, the optimized system model allows for producing an environmentally efficient biofuel in comparison to fossil fuel and bioethanol from sugar cane.

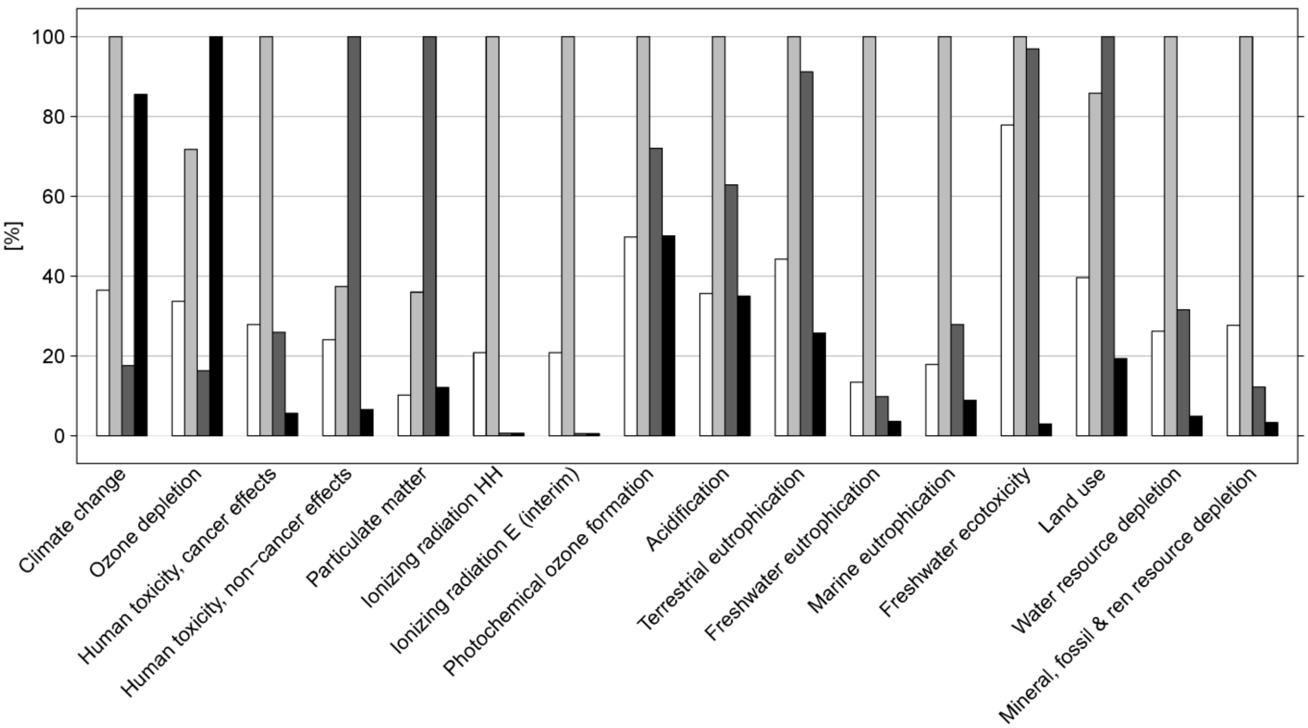

Figure 2. Life cycle impact assessment of 1 MJ obtained by combustion of petrol (black) and bioethanol from sugar cane (dark gray), from algae with the reference scenario (gray), and from algae with the optimized scenario (white).

4. Discussion

Co-product handling is a reiterated topic in LCA. We handled the obtained co-product, which was valorized as animal feedstock as for fermentation residues issued from other starch-rich biomasses (potatoes, corn, rye) (Jungbluth et al. 2007), by energy allocation as recommended by the European Commission for biofuels (European Commission 2009). As a consequence, only a quarter of the environmental burdens of the system were attributed to the produced bioethanol. Following ISO 14044 (2006), allocation should, however, be avoided wherever possible by dividing the unit process to be allocated or by expanding the product system. In order to avoid allocation, we extended the product system by substituting the produced co-product (DDGS equivalent, valorized as animal feedstock) for animal feedstocks, such as DDGS from rye, or grass silage (from intensive farming). Substitution of animal feeds is based on the dry weight of the feedstocks and their neutral detergent fiber (NDF) content, as NDF is the most common measure of fiber used for animal feed analysis. The composition of DDGS from rye was taken from the animal feed resources information system Feedipedia (www.feedipedia.com) and the one of grass silage from Yan and Agnew (2004). For the production of 1 kg ethanol from *Ulva*, 6.24 kg DDGS equivalent from *Ulva* (fresh weight) replace 5.85 kg of DDGS from rye or 10.11 kg grass silage. Whether a DDGS equivalent co-product from *Ulva* grown on intensive fish farming effluents is suitable as animal feed, due to (1) potential contamination with pharmaceuticals and other pollutants from the fish farm absorbed by the macroalgae and (2) possible aversion of animals to DDGS equivalent feed, remains, however, to be studied and discussed. The goal of comparing the different co-product handlings was to evaluate the impact of the choice of co-product handling on LCA results.

Handling the co-product by substitution instead of energy allocation significantly reduced the environmental impacts of the system and resulted in environmental bonuses in several impact categories for both replaced animal feedstocks (Figure 3). Thus, the management of the co-product in the LCA model (energy allocation vs. substitution) highly influenced the impact assessment results.

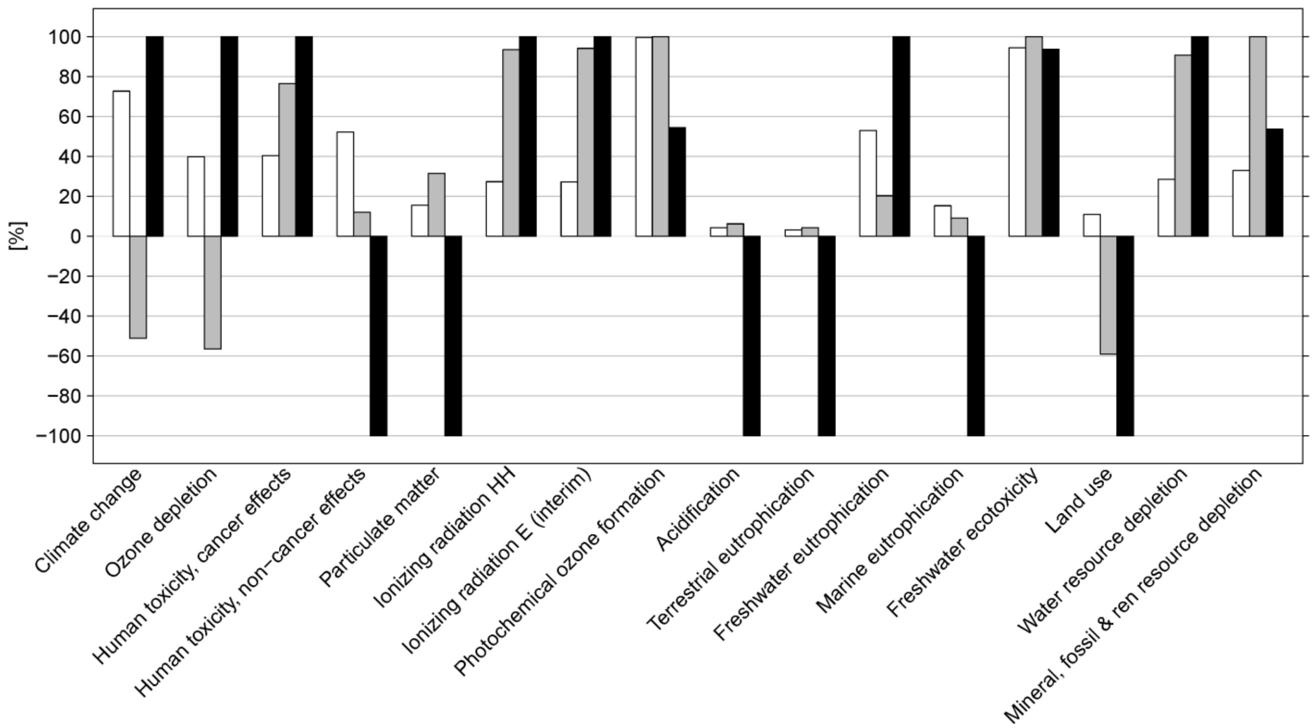

Figure 3. Life cycle impact assessment of one mega joule (MJ) obtained by combustion of bioethanol from algae with the optimized scenario applying energetic allocation (white), substitution for DDGS from rye (gray), and substitution for grass silage (black).

5. Conclusion

Based on literature data and preliminary studies, this work assesses environmental burdens of bioethanol from onshore grown macroalgae. The study revealed that an optimized system model allows for producing an environmentally efficient biofuel in comparison to fossil fuel and bioethanol from sugar cane. LCA results were highly dependent on the type of co-product management selected, as changing the co-product management from energy allocation to substitution significantly reduced the environmental burdens of the studied system.

6. Acknowledgements

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Development of Climate Choice Lunch concept for restaurants based on carbon footprinting

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ABSTRACT

A concept for communication of climate impacts of lunches was developed in Finland. The criteria for a Climate Choice Lunch concept were created in a stakeholder dialogue and three restaurant operators piloted the concept in 25 restaurants in spring 2014. The pilot was based on simplified carbon footprint assessment of 105 ordinary lunches, consisting of around 200 different ingredients. Quantitative assessment was made to understand variability and reasons of plate level climate impacts and to set up quantitative criteria, maximum limit, for a Climate Choice Lunch. Other sustainability criteria were also included in the concept. The pilot confirmed that to promote climate-friendly eating a long-term concept, instead of a short campaign, is needed. There is interest among consumers and restaurants for climate impact information on food. Lunch is regarded as a good opportunity for consumers to learn about climate-friendly eating. The main challenge is to produce sufficient reliable background data.

Keywords: carbon footprint, communication, consumer, eco-design, stakeholder dialogue

1. Introduction

A quarter of climate impacts of consumption come from food production and its consumption (Seppälä et al., 2011), yet consumers do not have enough information or sufficient understanding to make climate-friendly choices about food (Hartikainen et al. 2014). The most significant reductions in greenhouse gas emissions for food consumption can be made at the diet level. Reducing consumption of meat, eating seasonal foods and eating according to energy needs are key issues. Not all the responsibility can be placed on consumers, but they could play a significant role by making informed choices that are good for the environment and their health.

Consumer communication of climate impacts of foods should be made very simple, but challenges are substantial and consumers do not perceive food consumption as being a significant source of environmental impacts, particularly regarding contribution to greenhouse gas emissions (Hartikainen et al. 2014). Therefore, the Climate Lunch project (2013-2014) was established with two specific aims: to raise consumer awareness of climate impacts of food and to concretize the features of climate-friendly food portions and ingredients. The project aimed to decrease the climate impact of Finnish lunches significantly by offering a climate friendly lunch alternative to Finnish consumers, and indirectly by educating them about climate friendly food choices in general.

During the project, a concept of a Climate Choice Lunch was developed and tested in co-operation with pilot restaurants. The project defined, in close co-operation with stakeholders in the Finnish food sector, a set of criteria for the Climate Choice Lunch based on reliable and unbiased scientific information that are sufficiently ambitious enough in terms of reducing climate impact. This concept is significantly different to other campaigns on climate friendly eating in that the maximum limit of climate impact for Climate Choice and Better Climate Choice were defined and also other sustainability and nutritional criteria were considered. Efforts were also made to make sure that all the criteria would be acceptable to all stakeholders in the Finnish food sector.

Additionally, the concept of the Climate Choice Lunch aimed to be simple enough to be easily integrated into the current production systems of Finnish restaurants and cause minimum extra work to the restaurant staff. Developing the concept balancing between scientific precision, simple consumer communication and practicality for service providers has been crucial.

2. Defining criteria for Climate Choice Lunches in a stakeholder dialogue

The planning started with a review of initiatives, campaigns and concepts related to climate impacts of ready meals and lunches. Nothing similar to the planned concept was found, but different types of campaigns were found, including ‘Meatless Mondays’. In particular, no information was found on the successfulness or lessons learned from different initiatives. Closest to the planned concept was a project in Finland, in which one of the restaurants that participated in the Climate Lunch project had also participated a few years ago and in which a one-week climate friendly lunch campaign was staged. The recipes were not evaluated and the climate friendliness was only assumed by offering seasonal vegetables and vegetarian and fish meals.

A second review was made of methods to evaluate additional sustainability aspects of food. It was noted that no meal should be promoted that could be considered a non-sustainable choice in another impact category. Therefore, nutritional and other sustainability criteria were reviewed. Mainly issues of social responsibility, such as animal welfare, working conditions, product safety, and environmental impacts (eutrophication and acidification) were looked at, but we did not find enough information to make justified exclusions of specific ingredients. It was also noted that there are insufficient easily-available, science-based criteria for animal welfare, working conditions etc. It was also seen that animal welfare could conflict with climate impact. Product safety in Finland is already well developed and was not necessary to include as a criterion. It was also considered if fair trade products should be recommended, but it was not seen as a relevant criterion for climate friendly concept. Regarding other environmental impacts, such as eutrophication and acidification, there are not enough data in the literature to estimate them qualitatively.

Through a stakeholder dialogue with restaurants, government, health organizations, environmental organizations etc., consensus for inclusion of criteria for climate-friendly lunches other than for climate and carbon footprint maximum limit were set. First, results of the review were presented to the project's steering group. Afterwards a stakeholder workshop was held for 23 participants to comment on and accept the steering group's choices on criteria other than for climate and to define target maximum limit for climate impact of climate-friendly lunches. A simplified carbon footprint assessment formed the basis for the stakeholder discussions on the quantitative climate impact maximum limit of the Climate Choice Lunch concept. Voting on the target maximum CO_2-eq. limit, different stakeholders exhibited differing preferences on the ambitiousness. The restaurants and food industry were much more careful in setting the reduction target than were NGOs, which were more courageous. Ultimately, the project steering group suggested two levels for labeling, (standard) Climate Choice and Better Climate Choice. Two levels of labeling were seen as being more complex for consumers, but were acknowledged as being crucial to the concept, being both credible and ambitious enough in the eyes of environmentally aware persons while, being of general interest (for people who want to eat at least some meat). Milk and bread were always fixed in a meal according to national public mass catering guidelines and restaurants have only few possibilities to affect their climate impact. Therefore, the limits for the lunches were defined for the meal components for which restaurants can make significant changes to recipes and thus to climate impacts: main course, possible side dish (pasta, rice, potatoes etc.) and side salad.

Final decisions were made in the steering group after the workshop. The limit for the Climate Choice Lunch was defined as 15% less emissions than an average lunch, which means it can still include at least some meat, and for the Better Climate Choice Lunch at 30% less emissions than average. This means that the maximum climate impact of a Climate Choice Lunch is 0.65 kg CO_2-eq./main course and side salad, and for a Better Climate Choice 0.8 kg CO_2-eq./main course and side salad, based on the quantitative assessment of climate impact of lunches presented in chapter 3. Thus, the Better Climate Choice Lunches were almost all vegetarian, expect for some herring, pollock and salmon dishes. Climate Choice Lunches were also mainly vegetarian, with some fish dishes and a few pork or broiler dishes. Generally speaking meat dishes needed to have a side salad with a lower climate impact than average salad and relatively less meat and more vegetables to have a climate impact below the threshold.

Finally, WWF's sustainable fish list and national public catering nutritional recommendations applied as minimum criteria for the concept. The public recommendations define a nutritious meal and they include a glass of milk or sour milk with the meal. Because of the relatively high climate impact of animal products, the inclusion of milk was greatly discussed, but in the end a glass of milk was included in the concept in line with the recommendations.

3. Quantitative assessment of climate impacts of lunches

In the planning process of the concept it was already clear that greenhouse gas emissions for all ingredients from different production systems and countries, as used by the restaurants involved, could not be estimated in detail. Therefore, a simplified carbon footprint assessment was conducted based on MTT's previous Life-Cycle-Assessment (LCA) studies and literature reviews, and new scientific literature.

There was no intention to study the precise greenhouse gas emissions of a specific lunch meal based on particular LCA guidelines, but the order of magnitude of different types of lunches and the contribution of different ingredients to total climate impacts of meals was sought. The estimation would serve at this point only to enable development of the Climate Choice Lunch concept, its criteria and incentives, and piloting of the concept for one week.

The main criterion of the concept being climate impact, it was crucial that it be based on as reliable background information as possible. However, it is clear that the comparability of climate impact results from different studies and different products are not directly comparable. The lack of comparability and harmonization creates challenges and limitations for communicating initiatives such as this concept. The best available and applicable data from previous carbon footprinting projects were used, a few data gaps were filled and where regarded as important, a few production systems and average regional and or seasonally adopted estimates were created (such as for tomato, cucumber and salad production in south, central and northern Europe in winter).

Impacts of 105 lunches from three types of lunch restaurants were estimated, comprising around 200 different ingredients. The emissions were estimated for each restaurant's one-week menu and mainly for the raw-material production of ingredients in the recipes. They could be unprocessed raw materials, such as uncooked and unpeeled potatoes, or readily processed and cooked ingredients, such as fish fingers.

As the major share of a meal's climate impact comes from its primary production and due to limited resources of the project, mainly emission figures readily available in LCA literature, such as those from agricultural stage and input production (energy, fertilizers etc.), were included in the assessment.

As figures for processing, transport and storage were not always included in the studies available, they were not systematically included. In the case of processed ingredients, the original recipes were assessed so as to assess the actual amount of raw materials used (wheat, sugar, vegetable oil etc.). Generally, no emissions from the processing stage were included. Production losses were not taken into consideration systematically, even though they can have significant influence. Yet, because of its significance, values for the yield of processing stage of meat were included, and thus boneless meat was considered in recipes. Production losses when cooking meat (water loss) were also considered when restaurants used cooked ingredients in their recipes. In addition, it was known that emissions from processing and cooking in restaurants would be significant, but the restaurant sector does not currently record enough information on the corresponding energy consumption, which would have been allocated differently for different lunch meals.

The estimation was not made according to real consumption data of consumers in different restaurants, but to allow comparisons of different meals from different restaurants, estimations were in line with the standardized meal composition as defined in the Finnish public catering nutritional recommendations (Ministry of Social Affairs and Heath 2010). Therefore, a meal included a main course (400g of soup, 300g of casserole, 150 g of meal sauce or 120 g of meatballs or fish or equivalent + 30 g of sauce), with a side dish (100 g rice, 150 g potatoes or 120 g pasta, all weights as boiled), 200 g of side salad, 30 g slice of bread (or two if main course was soup), 5 g of margarine per slice of bread and 170 g of milk. The recommendations also defined minimum fiber and vegetable intake per meal, and maximum total and saturated fat and salt intake per meal.

The climate impacts of complete meals ranged between 0.6-2.8 kg CO2-eq. per meal, the average being 1.21 kg CO2-eq. per meal or 0.95 kg CO2-eq. per main course and side salad (some examples are given in Figure 1). According to the results, the climate impacts of main courses and side salads varied greatly. The largest impact of a main course was almost 20-fold compared with the smallest. The largest burden of a side salad was fourfold in comparison with the salad with the smallest burden. On average, a main course caused 45% of the impact of a meal, a side salad almost 30%, milk 20% and bread less than 10%. According to the results, differences in different lunch plates were large. By composing a meal differently, or even only changing the recipe of the main dish or side salad, climate impact could be decreased significantly without compromising nutrition.

Vegetarian meals usually had significantly lower emissions than the average meal. However, the climate impact of some meals that included a lot of cheese, cream or northern European vegetables grown in greenhouses during winter, were above the average. Fish main courses generally had low impacts, except for a few salmon dishes. Whether a fish meal was below or over the average depended greatly on the impacts of the side salad. The emissions of meat dishes were at or above the average. Moderate meat consumption as a part of well composed meal can keep the burden to the average level. For example, a pork stew with a side salad, which has low climate impact, can be below the average. Also the burden of pasta with mincemeat sauce can be average or very high depending on the type and amount of meat.

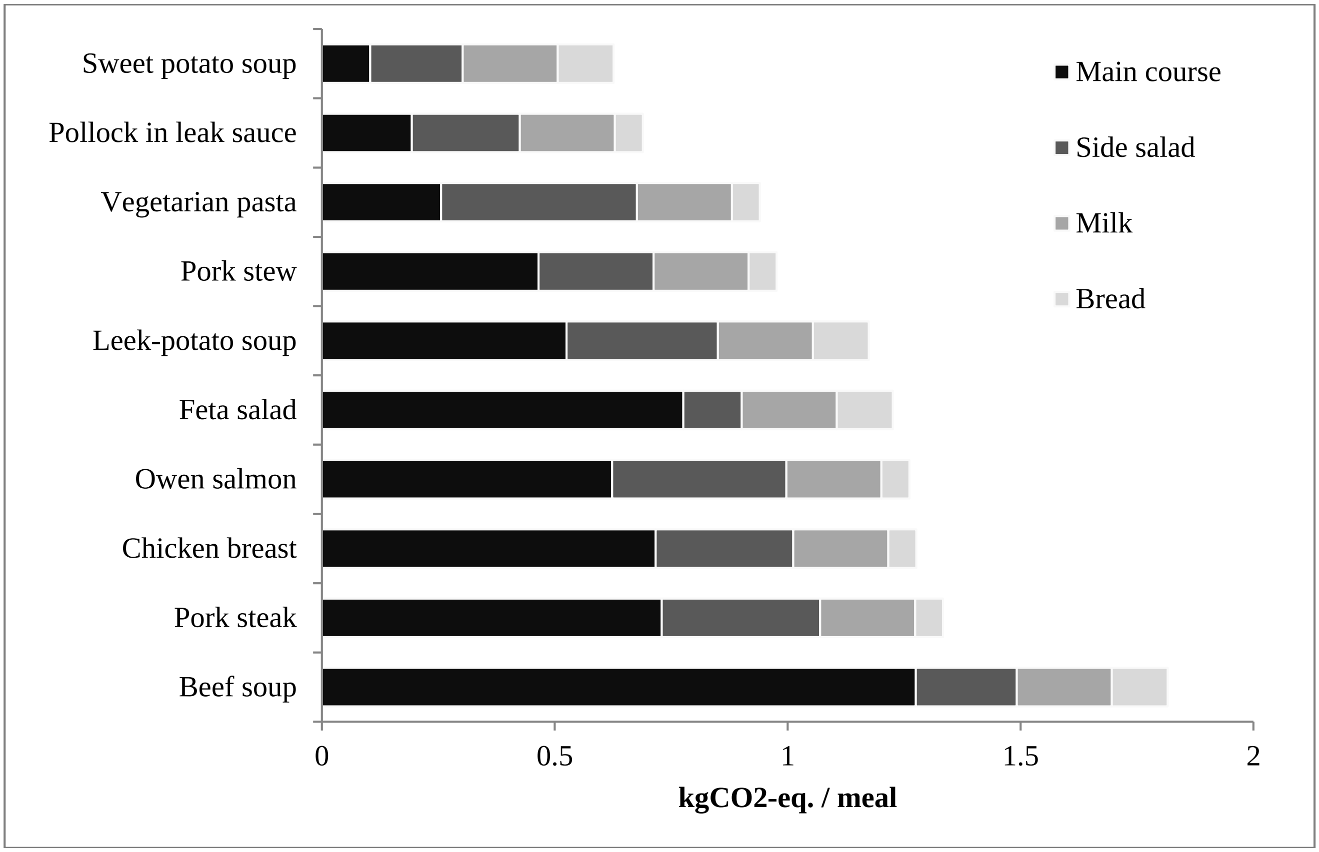

Figure 1. Examples of climate impacts of lunches.

4. Piloting the Climate Choice Lunch concept

To ensure practicability and to pilot the developed concept, three types of restaurants were included in the process. All restaurants offered a buffet lunch with 3-6 options per day, including at least one vegetarian option. Two of the 25 restaurants were office canteens, on the premises of an industrial company and one a canteen of an environmental institute. One restaurant was a public canteen for mainly office workers and students and this was the only one outside the main metropolitan area of Helsinki. The three were part of the leading Nordic catering company. One restaurant was a high quality but small restaurant of a catering school already committed to the Nordic Swan environmental label. The remaining 21 restaurants belonged to the same chain of canteens for students and offered government subsidized lunches.

The restaurants offered the project lunch recipes over a typical week. Based on the initial assessment, most of the restaurants needed to make changes in their recipes to lower climate impacts of their meals and to have Climate Choice and Better Climate Choice lunches available on their menus every day. Restaurants planned their own communications (both on-site and on the Internet) of the pilot weeks based on a common layout, logo (see Figure 2) and slogan developed in the project. After the pilot stage, restaurant staff gave valuable feedback on the practicality of the concept.

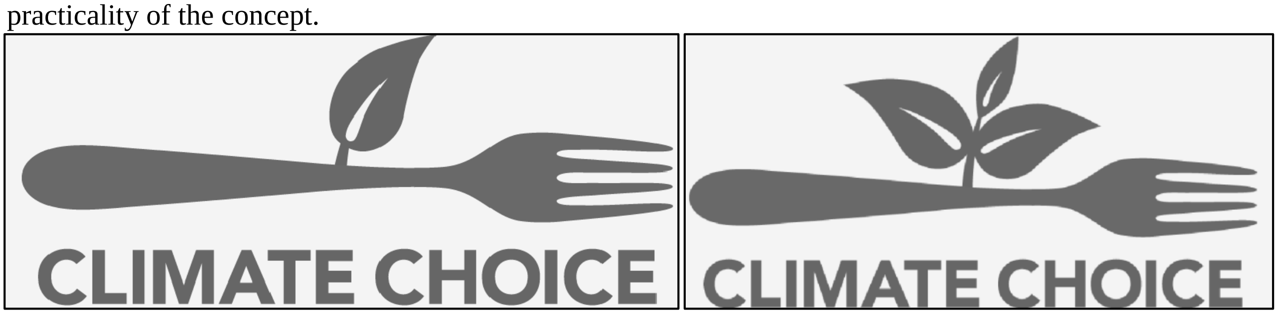

Figure 2. Logos for Climate Choice (one leaf) and Better Climate Choice (three leaves) lunches.

The starting point of the project was that most consumers do not have sufficient understanding to make informed climate-friendly decisions, which has been discovered in the earlier projects of MTT. Therefore, simple, label-based communication was considered the best way to communicate the issue to most consumers. Most of the restaurants differentiated Climate Choice meals from other meals by labeling them in the buffet. Only one restaurant marked Climate Choice Lunches solely on the menu available at the buffet. In addition, some restaurants informed about the Climate Choice Lunch week on tables where clients ate and in posters around the restaurant. Some restaurants also labeled their menus on the Internet.

The aim of the consumer studies was to have information on consumers' first impressions of the concept: if they liked the idea or not, why they chose Climate Choice Lunch if they did and whether they would choose it again. Consumer studies were conducted in two simple ways. A one page paper survey was available to be filled in voluntarily in restaurants on Thursdays or Fridays, after four or five days of the pilot phase. In total 307 paper responses were received. In addition, on Wednesdays and Thursdays of the pilot week, 33 interviews were carried out in 4 different restaurants. A short electronic survey was conducted also among restaurant staff about their experiences on the concept and the pilot week.

The concept was tested in 25 restaurants in spring 2014. The motivation of restaurant staff to ensure survey responses from consumers was very important, and in this project it was successful in some restaurants but not all. Therefore, not as many responses were received as hoped for.

Based on the 33 interviews, only around 50% of customers seemed to notice the pilot of Climate Choice Lunch. Information given on the tables where people were eating seemed to be the most efficient means of communication and that information was registered best when put on tables two weeks before the pilot week. A few consumers actually stated they were too hungry and busy when they are queuing to notice information at that stage. Labels that were placed on the buffet, where people choose their main course and which was thought to be the most noticeable and important site for communication, were actually noticed the least. Some people noticed the labels on the menus before the buffet. Very few customers noticed that there were two levels of Climate Choices. Customers also automatically thought that all vegetarian meals would be climate friendly: many people that had chosen the vegetarian meal thought that they would have opted for the Climate Choice, even if that was not true.

It seemed that during a short campaign as represented by this pilot phase, people who reacted quickly to communication (here the label) and chose the climate-friendly lunch, did so because they already thought about the environment when making decisions on consumption. Few customers acknowledged that the idea of thinking about climate impacts during lunch was new to them, even though they might have thought about it when grocery shopping. Consumers who had noticed the campaign, and were probably more environmentally aware, wanted more information on why specific lunches were Climate Choices. Even if the sales of Climate Choice meals did not increase compared with a reference week, consumers felt positive towards the concept, and thought it gave them information in a very simple way and which they could use to make better choices.

The 307 paper responses indicated that the most important criteria for choosing a meal in general were the attractiveness and expected taste of the dish. It was clear that consumers, who chose a Climate Choice Lunch, chose it because of expected healthiness of the meal more often than for environmental reasons. Even though the criteria of Climate Choice Lunches included health criteria, it should not be expected that consumers had learned that in a one-week period. 40% of the 307 respondents in the paper survey stated that they would choose a Climate Choice Lunch at least often, if not always. 54% stated they would choose it at least every now and then.

Restaurant staff seemed to feel very positive about the new concept. Extra work is needed mainly from the staff that plan the new recipes, but others did not seem to mind the little extra work as most were interested in and felt engaged with the concept.

4. Discussion

Even though an approximate climate impact assessment of ingredients was made, it is apparent that more detailed and comprehensive carbon footprint databases are needed for restaurants to evaluate and design lunches. Creating a reliable and harmonized database for a credible concept remains a challenge.

Some secondary data can be collected with rather limited resources from the literature and databases but validating, harmonizing, filling the data gaps and reporting data requires multiple efforts. 50% of the lunches varied between 0.94 and 1.37 kg CO2-eq./meal. The range shows that the definition for whether a lunch is climate-friendly or not is very sensitive to even small changes in the amounts of ingredients or changes in the ingredients themselves. Therefore, it is crucial that a credible future concept relies on good quality carbon footprint data based on scientific and objective research.

Not only harmonization of methodologies used in different studies, such as allocation and consideration of local production circumstances, including soil types, but also system boundaries and practical calculation procedures (e.g. energy consumption of field machinery) need to be harmonized. Also more life cycle stages, such as processing and freezing and in particular raw material losses, should be added. Uncertainties and variability should be assessed to make reliable differentiations between climate-friendly and other lunches. In addition, data have to be transparently reported provided to any stakeholder who wishes to verify them. Only then a future low-carbon meal concept can be reliable. This kind of research should be conducted objectively, but with a focus on the largest contributions and the most uncertain estimates so as not to be too overwhelming.

Recipes and menus change often and in the future databases for climate impacts of ingredients and processing should be integrated with restaurants' recipe software. Such software already assesses, for example, nutritional values and it would be most practical for restaurants that the same program would assess climate impacts. Hence, restaurants could use such a program as an eco-design tool for meals and could verify their communication with it.

It would be good to include energy consumption for cooking and losses during processing in a concept like this because it could provide relatively interesting information for consumers when choosing their lunch restaurant or for differentiating between differently cooked meals. It would however require that more information on energy consumption and losses from food processing would be available to enable fair comparisons to be made among restaurants using different shares of raw materials (such as vegetables) and processed ingredients (vegetable steaks), which consume energy and create waste streams during different stages. According to one small Finnish study including very few restaurants, the energy and heat consumption of different restaurants can vary between 0.5-2.3 kWh/meal. Using average Finnish electricity and heat production emission factors, the greenhouse gas emissions would vary between 0.2-0.6 kg CO_2-eq. per meal. The largest energy consumption was measured for an à la carte restaurant, and the smallest one in a school canteen. For an average public restaurant adopting the Climate Choice Lunch concept we could estimate the value to be around the average. As losses can be significant and directly cause direct increases in emissions during raw material production, it appears that the impacts of both energy consumption and losses are significant at the meal level and should be taken into consideration in the future in developing such concepts, but for the time being it is not feasible for restaurants to collect the necessary data nor to differentiate among meals.

The consumer study results should be considered in the light of this kind of concept being unlikely to succeed as a short campaign, but perhaps being much more successful as a permanent feature of a restaurant. Therefore, the learning aspect would increase slowly and people would come to understand more about the information provided. Currently the two levels of Climate Choice Lunches were not recognized as wanted. Moreover, some of the customers who showed interest in the concept regarded the simple information as deficient, and they would have preferred more detailed information.

It is hard to communicate climate impacts of lunches to consumers when most consumers do not consider food and agriculture as being significant sources of environmental impacts, not to speak of climate impact. Even when environmentally aware consumers think about environmental impacts of foods, they might think about them when they are grocery shopping for home cooking, but not when eating lunch in a restaurant. It is necessary to emphasize to the consumers that lunch choices can be a good and simple way to influence the climate through a concept like this. It is understandable that this kind of concept would take a longer period than a week to become noticed and increase understanding significantly, and subsequently affect consumer behavior.

It is obvious that most of those who would choose a Climate Choice Lunch currently would do it either for health reasons or because they are already vegetarians and would anyway choose a vegetarian meal for animal rights concerns. Therefore, emission reductions could also be achieved by educating restaurants in provision of climate-friendly vegetarian meals. Climate friendliness and healthiness could then go hand in hand, e.g. reducing the amount of cheese and cream consumed and increasing the amount of beans and lentils. However, to make vegetarian dishes more attractive to others than vegetarians, and to show them that vegetarian meals can be tasty, cream and cheese are often used. To be able to develop a concept where some animal-based products can be used and dishes made attractive to all, climate impacts need to be estimated with good reliability rather than be based on a simple list of high and low impact ingredients, which would lead to vegetarian meals being the sole climate-friendly options.

As some consumers said, information on climate impacts of meals can have an impact on the buying decision when two alternatives are otherwise equal. Therefore, climate-friendly lunches have to be made attractive, in particular, if they do not contain meat.

5. Conclusion

Consumers need to be given simple ways to affect the climate burden of their consumption. Small changes, like choosing one of the two Climate Choice Lunches a couple of times a week, can directly decrease climate impacts of food consumption significantly. In addition, consumers can start to learn slowly what climate-friendly eating actually means. The pilot project confirmed the idea that promoting climate-friendly eating in a campaign is not enough, a long-term concept is needed.

Lunch represents a challenge, but if successful could represent a very successful opportunity to communicate climate impacts of foods to consumers. It is hard to make people notice communications in a restaurant when they come there hungry and busy, but as so many people eat lunch in a canteen every day, it could represent a very efficient way to inform consumers as they are likely to be open for new information in canteens. The first step would still be to make ordinary consumers think and talk about climate change and to bring the subject closer to consumers.

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A modification of supply chain of green bean in Indonesia on basis of LCA thinking

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ABSTRACT

Climate change is mainly linked to greenhouse gas (GHG) emissions in which the agricultural sector occupies 14% of total emissions. In this paper, the questionnaires were implemented to investigate the effects of green bean quality including eco-burden factor and price on consumer buying decision. Also, on the estimation of eco-burden, LCA methodology was considered, and the carbon footprint of green bean in the supply chain process in Indonesia was expressed. The results showed that the total emissions (CFP) of green bean were between 4.92 and 7.38 kg-CO_2eq/kg green bean by varying farmers, and they became larger than that of Japan case (1.11 kg-CO_2eq/kg green bean). In addition, through our questionnaires on basis of the quality and price of green bean, we confirmed that the factor of quality is more significant for consumer buying decision.

Keywords: LCI, green bean, supply chain, carbon footprint of product, quality change

1. Introduction

Food sustainability is unquestionably a major issue for the years to come (Catherine et al., 2013). Nevertheless, the food industry is one of the world's largest industrial sectors and a large user of energy. Food production, preservation and distribution consume a considerable amount of energy, which contributes to total CO_2 emission. Andersson et al. (1994) stated, there is an increasing awareness that the environmentally conscious consumers of the future will consider ecological and ethical criteria in selecting food products. In addition, Boer (2002) reported, consumers in developed countries demand safe food of high quality that has been produced with minimal adverse impacts on the environment. It is thus essential to evaluate the environmental impact and the utilization of resources in food production and distribution systems for sustainable consumption (Poritosh et al., 2009).

Based on NASA-Goddard Institute for Space Studies (2012), the climate changes are mainly linked to GHG emissions in which the processes of farming, land use change and transport contribute 14%, 18% and 14% of total emissions. Thus, the reduction of GHG emissions for agriculture and food systems would be required (Catherine et al., 2013). For instance, the improvement of food supply chain would be brought by the higher efficiency of energy usage, and that would contribute to the reduction of GHG emissions.

By mapping all ingredients and components contained in the product, it will be able to assess the environmental impact (Alan et al., 2010). The application of supply chain management techniques has proven to dramatically improve efficiencies in a variety of industries (Haarteveit et al., 2004). As a more systematic and integrated strategy, the green supply chain management has emerged as an important new innovation. This fact helps farmers to develop strategies so as to earn profit and market share by lowering their environmental risks and impacts, while raising their ecological efficiencies (Qinghua et al., 2007). For that reason, our purpose is to evaluate carbon footprint of fresh and rejected green bean along its supply chain in Indonesia. Also, we argued the effect of CO_2 emissions are reflected to the product quality.

In previous studies, it was confirmed that eco-labeling was a way to encourage consumers to prefer the products whose characteristics are more environmentally friendly (Ralph et al., 2007). In the consumer research carried out in recent years, it was shown that consumers would be affected by not only the demand change of reliable products, but also by the corporate social responsibility (i.e. environmental consciousness) to some extent (Turan, 2007). Also, in general, since vegetables are perishable, the growth of these agricultural products is affected by the capability and/or availability of supply chain so as to be quickly delivered to consumer with fresh condition (Madeleine and Zhaohui, 2012). Thus, since the quality of product is an important factor, we should consider the quality change during the handling process. As a first step, we focus on the comparatively popular agricultural product, and estimate the carbon footprint of product of it on basis of LCA methodology. We attempt to verify the relationship between the quality including eco-burden element and price for consumer buying decision. Here, the reason why we select the Indonesia area as a case study is as follows: the cold chain is not enough constructing, the economic loss with quality worse due to the proper climatic condition is caused and the residential people has a great concern on eco-friendliness. Moreover, based on our preliminary research, it shows that 75% of citizen in Indonesia often eat green bean within a week which means that it is one of the popular vegetable in Indonesia.

2. Methods

2.1. Basic supply chain of vegetable

Chien-Jung et al. (2013) pointed out that all supply chains cover the processes from the raw material supply to the added value providing, including the sale to customers. In this study, we focused on the Indonesia market. For the supply chain of vegetable in Indonesia, as illustrated in Figure 1, some processes of nursery, planting, growing and harvesting are done by farmers. Until the agricultural products are harvested, some of them are not harvested and are consciously left in the field for new seed production. Once vegetables are harvested, they are transported in use of small trucks from the field to the warehouse managed by farmer union. If the transportation distance to the warehouse is less than 500 meters, they will be carried by man power. Otherwise, the truck is used and the harvested products are sorted and graded. The purpose for sorting and grading the products in the warehouse is to discriminate each product for whether it is suitable for export, modern or traditional market. After they are packed, these products are delivered directly to a distributor, by whom they are sold to consumer or just stored in their warehouse.

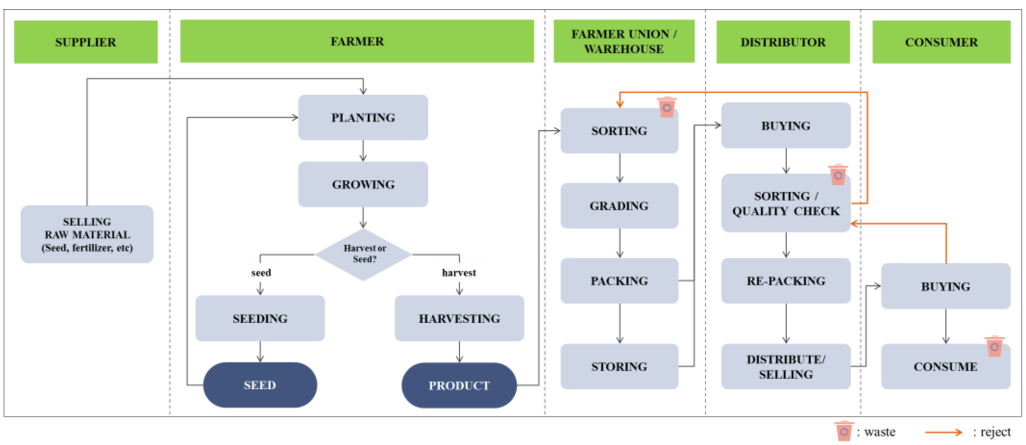

Figure 1. Basic supply chain of vegetable in Indonesia

Next, in a distributor, the agricultural products such as vegetables will be re-sorted on basis of quality requirement. This process is needed because the physical quality of them usually changes due to the influence of transportation process. As a result, all products are determined whether they have commercial value or not. The unfit products mean irregular and/or rotten ones. Those worse products would be caused due to their inappropriate cultivation or physical damages. Thus, both of them would be returned to the farmer, and the irregular product will be re-delivered to another consumer. While, the rotten ones will be no longer possible to be consumed. As Diana (2009) described about the related facts, if growers do not satisfy with the pre-harvest food safety standards, their crops would become waste as significant losses. Also, from the same viewpoint of Lundqvist et al. (2008), they defined that the food waste is the deliberately discarded food. It is no longer possible to eat for human consumption, even if its state would still be eatable. Recently, the requirements of end-consumers have been increasingly requested with regard to the better qualities of food products, assortment and package features, and the presented way of food products at a food store (Jon and Taras, 2009). Simultaneously, the loss percentage of agricultural products of vegetables is approximately 25% to 30% in the following processes of packing, transportation and storage (Dan, 2012). However, this percentage is very small for that of total waste due to its quality loss. Note that the terminology of "loss percentage" does not mean the amount with lost quality but the amount which accidentally fall off in the delivering process.

On the other hand, from the viewpoint of environmental impact, the impact for the lost amount in supply chain becomes smaller than that of the waste disposal, because the total lost amount in supply chain is less than 5% than the percentage of disposal amount including the factors of quality loss. If we describe about the carbon footprint of products and other environmental indexes for agricultural products, we have to discuss about not the material balance in the supply chain but the quality change of target product. Accordingly, in this paper, we focused on the agricultural product of green bean whose quality would be changed in some processes including the delivery process to more extent and whose supplied amount would be affected by its quality loss.

2.2. LCA methodology

LCA methodology is required for evaluating the carbon footprint of product in the supply chain process. Almudena et al. (2010) assessed the environmental impacts of products and services in use of LCA, which has been applied to many industrial sectors including food and agriculture. Guido et al. (2000) emphasize that the use of LCA in agricultural sector can cover all environmental impacts and that the result can provide us a significant potential for supporting the environmental improvement and innovation of the proposed agribusiness by Rene (Rene, 2002). This methodology is used to compare the primary energy requirement (Michael, 2005) and to analyze environmental impacts that can occur at all steps of the product chain (Gaudreault et al., 2009) by quantifying and evaluating the consumed resources and/or the emissions at all stages of its life cycle from the extraction of resources, through the production of materials and products parts, and the use of the product including its reuse and/or the recycle of final disposal (Hayo, 2004). In this respect, LCA approach is used to evaluate the impacts from the view point of consumers, which might be able to be reduced significantly on their food purchases (Niels et al., 2000).

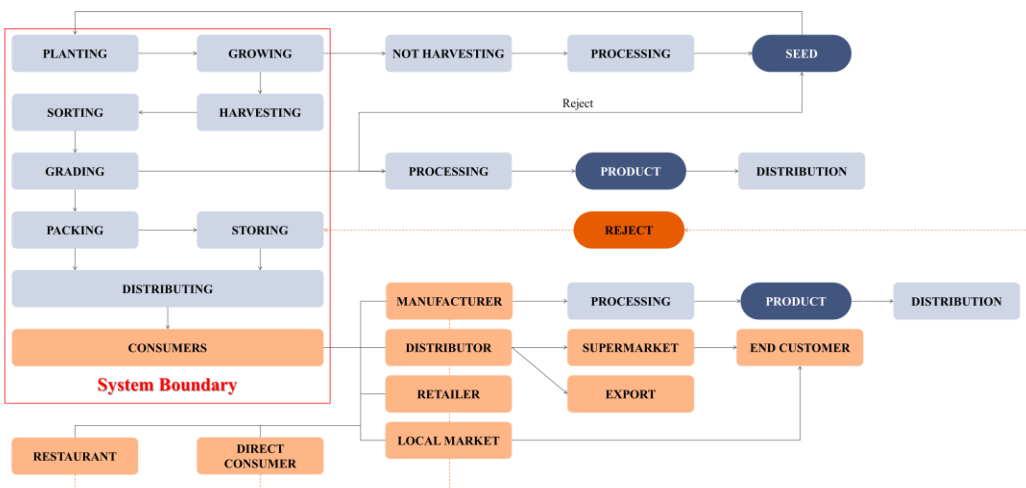

Figure 2. System boundary

In this study, as a first step, we estimated the LCI of green bean which harvested by the three different farmers. Actually, the production capacity would be varied by the cultivation area and/or farmers. Thus, considering these uncertainties, we analyzed the impacts. Also, we prepared the following two cases: (1) the LCI of fresh green bean and (2) that of rejected one. Here, the rejected green bean is defined as an irregular and/or a rotten product which are less commercial value. Note that the rate of rejected amount is from 13 to 20 % against the total harvesting product due to the difference of farmer (see Figure 3). The functional unit of LCI in this study is kg-CO_2eq/kg-green bean. Since the specific CO_2 emission of LCI includes other global warming gasses, that is evaluated as equivalent CO_2 emission. Also, assuming that the total equivalent CO_2 emission of i-th farmer for the final products, that for the fresh ones and that for the rejected ones are $LCI_{total,i}$, $LCI_{fresh,i}$ and $LCI_{rejected,i}$, respectively, $LCI_{total,i}$ is,

$$LCI_{total,i} = (1-\alpha)\, LCI_{fresh,i} + \alpha\, LCI_{rejected,i} \quad \text{Eq. 1.}$$

Where, α is the rate of rejected amount.

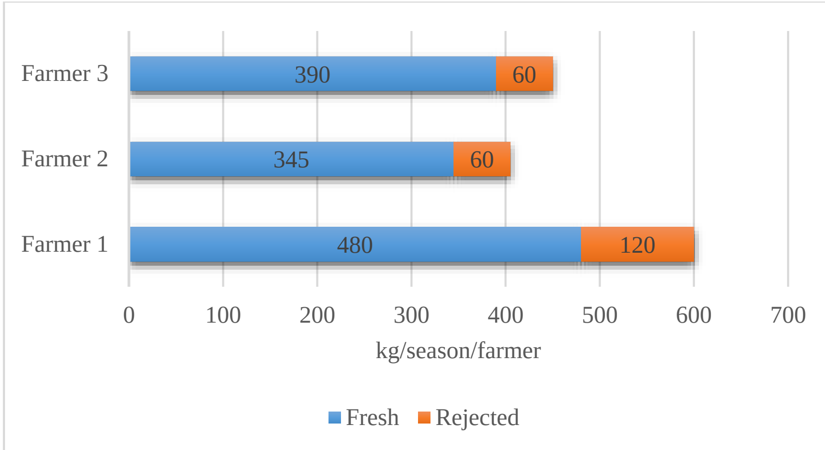

Figure 3. Comparison between fresh green bean and rejected one

Next, as shown in Figure 2, the system boundary of this study encompasses production (planting, growing and harvesting), transportation from field to warehouse, warehouse handling (sorting, grading, packing and storing), and transportation from warehouse to consumers. Based on these conditions, we estimated the LCI of green bean in consideration of the farmer difference and the harvested product difference due to the quality loss.

Here, Table 1 shows material inputs for each case of LCI calculation. For each case, we simulated that in use of LCA calculation software MiLCA (The Japan Environmental Management Association for Industry (JEMAI)). However, the loading ratio of truck long transport was assumed to be 62% of 15 ton truck due to the transportation condition of Japan since there is no data of that in the target area. The estimation of fuel consumption was based on the guideline of fuel consumption of truck (Ministry of Economy, Trade and Industry and Ministry of Land, Infrastructure, Transport and Tourism). Here, the fuel consumption of diesel truck is as follows:

$$LnF=2.71-0.812Ln(R/100)-0.656LnM \qquad \text{Eq. 2}$$

Where, F, R and M are the fuel consumption per t-km [L/tkm], the loading ratio [%] and maximum capacity [kg], respectively. Also, Ln means logarithm natural.

Table 1. Material inputs

Case	**Process**	**Material input**	**Farmer 1**	**Farmer 2**	**Farmer 3**
				Amount	
Fresh and Rejected	Production	Fertilizers	0.041 kg/kg	0.062 kg/kg	0.03 kg/kg
		Pesticide	0.063 kg/kg	0.095 kg/kg	0.1 kg/kg
		Herbicide	0.033 kg/kg	0.05 kg/kg	0.03 kg/kg
		Gasoline	1.54 L/kg	2.31 L/kg	2.1 L/kg
	Transportation from field to warehouse	-	1.5 ton truck	Man power	1.5 ton truck
		-	loading ratio 40%		loading ratio 30%
		-	distance 5 km	distance 0.5 km	distance 4 km
		Diesel	2.17 mL/kg	-	2.93 mL/kg
	Warehouse handling	Electricity	0.018 kWh/kg	0.027 kWh/kg	0.024 kWh/kg
	Transportation from warehouse to distributor	-	15 ton truck	15 ton truck	15 ton truck
		-	loading ratio 62%	loading ratio 62%	loading ratio 62%
		-	distance 32 km	distance 32 km	distance 40 km
		Diesel	0.14 mL/kg	0.14 mL/kg	0.17 mL/kg
Rejected	Transportation from distributor to warehouse	-	15 ton truck	15 ton truck	15 ton truck
		-	loading ratio 62%	loading ratio 62%	loading ratio 62%
		-	distance 32 km	distance 32 km	distance 40 km
		Diesel	0.14 mL/kg	0.14 mL/kg	0.17 mL/kg
	Warehouse handling	Electricity	0.092 kWh/kg	0.183 kWh/kg	0.183 kWh/kg

2.3. Consumer behavior analysis

Next, we argued about the following hypothesis: "Consumer buying decision on green bean would be influenced by variable quality and price". In the quality, the environmental factor is included, too. We perceived the relation between both variables at the time when consumer bought green bean on basis of a multiple regression analysis. This analysis is a statistical technique which can be used to analyze the linear relationship of $Y_1 = X_1 + X_2 + \ldots + X_n$. The objective of multiple regression analysis is to use the independent variables by which the selected single dependent value is predicted (Joseph et al., 2010). The dependent variable in this study is consumer buying decision (Y) and independent variables are quality (X_1) and price (X_2) of green bean. Also, we focused on consumer's behavior analysis so as to discover the importance of quality of green bean when the consumer buys it. For instance, according to the data of Ministry of Agriculture Indonesia, the average consumption weight of green bean in 2009 and 2010 were 832 g/capita and increased to 884 g/capita. Also, FAO reported the average times of eating green bean in Indonesia was 5 days per week with the total consumption of 3 g/capita/day. Thus, from the viewpoint of food LCA concept, on the often eaten food, to analyze the LCI and investigate the number of eating the product are significantly important. Note that the LCI would be affected by the qualities which are the factors of size, color, texture, flavor and nutrition content, and that the inventory result has to be based on the dynamic aspect. This time, we only referred to the LCI of green bean in consideration of the material balance.

In this study, the questionnaires were executed for a month between Jan 1 and 27, 2014 through the online system, and we collected 174 valid respondents. The investigated areas were Jakarta, Surabaya and Bandung in Indonesia. The questionnaire consists of the following 3 sections: (1) the respondent demographics profile, (2) the consumers' awareness on the environment and their health conditions, and (3) the measurement on important factors for consumers.

3. Results

3.1. Carbon footprint of product (CFP)

We used two sources to evaluate the carbon footprint of fresh or rejected green bean in consideration of material balance in the supply chain. These data were collected by the field investigation including the interview with farmers, and the other was based on the reference papers. As we mentioned above, we evaluated the specific CO_2 emissions of fresh green bean and rejected product. The initial green bean is categorized as follows: size of 8-12 cm, weight of 3-5 g, color of green, firm texture and unbruised bean. Here, both of fresh and rejected green beans are assumed to be transported to the distributor warehouse. Then, the rejected one is returned to the farmer.

Using the software of MiLCA in which approximately 3,000 inventories of materials, energy use and transportation were bundled (Haruhiro et al., 2013), we estimated the specific CO_2 emission on basis of Eq. 1. In each case, we obtained the CFP between 4.92 and 7.38 kg-CO_2eq/kg green bean. The influences of rejected green bean were from 0.94 to 1.15 kg-CO_2eq/kg green bean (see Figure 4). Compared to the CFP in the Japan case, there seems to be enough potential to reduce them of Indonesia to more extent. That is, according to Ajinomoto Group Environmental Report 2006, that of same product was 1.11 kg-CO_2eq/kg green bean, and that was equivalent to about 6 times against the CFP in this study.

Table 2. CFP of green bean supply chain in Indonesia (kg-CO_2eq/kg green bean)

	Total Global Warming	Production	Transportation	Warehouse handling
Farmer 1	4.92	4.88	0.02	0.02
Farmer 2	7.38	7.33	0.01	0.04
Farmer 3	6.86	6.80	0.02	0.04

According to Figure 4, the CO_2 emission of rejected product became larger in comparison to the fresh, since that has longer processes in the supply chain process. As illustrated in Figure 2, the rejected product is returned to the warehouse, and then, that is re-delivered to another consumer. That is, the added process requires more fossil fuel for the transportation and electricity in storage of product. The largest CO_2 emission in the production process was emitted in comparison to other ones in the green bean supply chain. This reason is due to the machineries utilization which is fueled by much gasoline.

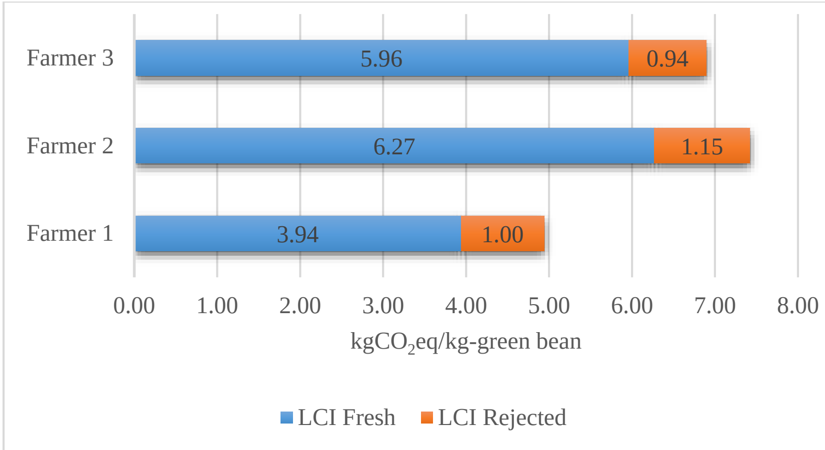

Figure 4. Comparison of emissions produced by fresh and rejected green bean

3.2. Consumer's behavior

This paper also has aim to analyze the relationship between quality and price at the time of purchase. As the result on basis of questionnaire, from first section of questionnaire a respondents profile was obtained. Table 3 shows the demographics profile of the respondents.

Table 3. Respondents' demographics profile

Category		Frequency	Percentage
City	Jakarta	92	52.87
	Surabaya	33	18.97
	Bandung	49	28.16
Gender	Male	78	44.83
	Female	96	55.17
Marital	Single	69	39.66
	Married	105	60.34
Education	Secondary	12	6.90
	Diploma	9	5.17
	Bachelor	105	60.34
	Postgraduate	48	27.59
Monthly income	Below USD 85	10	5.75
	USD 85 – USD 339	52	29.89
	USD 340 – USD 595	35	20.11
	USD 596 – USD 850	27	15.52
	Over USD 850	50	28.74

Also, we asked the respondents about their consciousness on environment and their health. More than half of respondents (66.67%) always dispose their garbage in the appropriate place, the 29.89% of them often do that, and the rest of them (3.45%) seldom do that in the appropriate place. That is, they have seems to have highly environmental awareness.

Table 4. Consumer's behavior

Questions	Never		Seldom		Often		Always	
	Total	%	Total	%	Total	%	Total	%
How often do you throw away your garbage in the appropriate place?	0	0.00	6	3.45	52	29.89	116	66.67
How often do you pay attention on your food consumption?	1	0.57	32	18.39	97	55.75	44	25.29
How often do you consume organic vegetables or fruits?	7	4.02	98	56.32	59	33.91	10	5.75
How many times do you shop for vegetables in one week?	17	9.77	47	27.01	73	41.95	37	21.26
How often do you consume vegetables in one week?	0	0.00	31	17.82	91	52.30	52	29.89

Next, in order to discover consumer's behavior in consuming vegetable, we asked the respondents about the importance of their food consumption. The majority people have an attention to the products (55.75%), the 25.29% always worry about the qualities. Although there are respondents who seldom do it (18.39%) or have no attention to it (0.57%), most consumers concern with their health conditions which reflected by their behavior in consuming of vegetables. On the other hand, on their consumptions status of both organic vegetable and fruit, almost respondents hesitate to consume them (56.32%). That is, it is shown that the organic products are unpopular for Indonesian.

On the behavior of purchases of vegetables, the 41.95% of respondents go shopping of them by a few times for a week. However, there are also respondents who seldom do it (27.01%) or do not go shopping (9.77%).

Finally, according to the personal habit of vegetables eating, the 52.30% of respondents often eat them per a week, or the 17.82% seldom do it. However, there is no people who do not eat them. That is, this results indicate that they are likely to eat vegetables for their health condition.

Using the questionnaire results, we analyzed them due to Likert-scale (i.e. 1=strongly disagree, 5=strongly agree) for estimating the statistical perception.

A multiple regression analysis was used to analyze the data. A validity test was conducted to evaluate the questionnaire. A questionnaire can be stated as valid if its questions were able to reveal something that will be measured by it. Using the correlation coefficient of Pearson, the results are clearly valid, because R-value is larger than sample correlation coefficient r table (0.148) and P-value is smaller than 0.05. Also, the t-values of quality (X_1) and price (X_2) are 8.15 and 2.10, respectively (See Table 5).

$$Y = 6.28 + 0.39 X_1 + 0.26 X_2 \quad \text{Eq. 3}$$

Based on this equation, the regression coefficient of quality was 0.39. This means that the quality would have a positive effect on consumer buying decision. For instance, if the quality of green bean is good, the consumers' awareness for buying that would increase. On the other hand, the variable of price was 0.26. That is, the consumers have awareness of the price to some extent, but, they have stronger interest in the condition of green bean at their purchases.

Table 5. Results of multiple regression analysis

Model Summary[b]

Case	Model	R	R square	Adjusted R square	Std. Error of the estimate
Green bean	1	0.61	0.37	0.36	2.17

a Predictors: (Constant), Quality, Price b Dependent Variable: Consumer decision

ANOVA[b]

Case	Model		Sum of squares	df	Mean square	F	Sig.
Green bean	1	Regression	477.16	2	238.58	50.55	0.000 (a)
		Residual	807.14	171	4.72		
		Total	1,284.30	173			

a Predictors: (Constant), Quality, Price b Dependent Variable: Consumer decision

Coefficients[a]

Case	Model		Unstandardized coefficients		Standardized coefficients	t
			B	Std. Error	Beta	
Green bean	1	(Constant)	6.28	0.93		6.76
		Quality	0.39	0.05		8.15
		Price	0.26	0.12		2.10

a Dependent Variable: Consumer decision

Coefficient of Correlations[a]

Case	Model			Quality	Price	Consumer' decision
Green bean	1	Correlations	Quality	14.16		
			Price	2.23	2.16	
			Consumer decision	6.10	1.43	7.38

4. Discussion

Based on our results, it was shown that the total emission of fresh green bean was smaller than that of the rejected. This implies the following matters: if the farmer who cultivates green bean does not pay attention to the handling of product, the rejected amount, that is, the waste amount would be increased. Likewise, in Indonesia where the annual consumption of green bean is large, the CO_2 emission for the target product might be attributed to the large amount of emission. Also, as we referred to the section of 3.1, the specific CO_2 emission of green bean was about 6 times larger than that of product of Japan. We have to think what kind of good countermeasure for reducing that we can propose. For instance, the Photovoltaic (PV) technology would be able to be promoted in the agricultural product. In the previous study by Eidelweijs et al. (2012), it was shown that the promotion of PV system on paprika distribution process in Indonesia can decrease CO_2 emission by 1.947 g-CO_2/paprika, that is, the 11.9% of cultivation process, the 30% of packaging, and the 30% of cold storage in comparison to the conventional case, respectively. As the same as the study by Fukumoto et al. (2011), the advanced paprika cultivation system in consideration of the energy supply of low carbon emission fuel such as a bio-fuel was proposed. Based on the biomass gasification process of Blue-Tower (BT) technology, they showed that the promotion of new technology can reduce the CO_2 emission on LCA concept. The BT co-generation (BT-CGS) was able to decrease by 104.6 gCO_2/paprika, that is, the CO_2 abatement was 82.0% in comparison to the conventional system.

According to Douadia and Pierre (2009), the shopping choices can contribute to the global warming protection. All over the world, the use of eco-labelling as a market-driven environmental policy tool is increasing. A number of articles have dealt with eliciting consumer's preferences and willingness to pay (WTP) for eco-friendly food products. Also, in the previous study about consumers' awareness in Jakarta, Indonesia, we understood that the consumers have a high awareness on global warming issues. Simultaneously, we knew that CO_2 emission factor is the most important factor and that their decision on purchase of paprika is affected by the communication index of eco-label. In other words, there would be somewhat potential market for paprika with eco-label in Indonesia. In that case, the WTP of consumers was corresponding to the 15% or more against the average existing price of paprika (Eidelweijs et al., 2012).

Furthermore, it was shown that the carbon footprint of product (CFP) for agriculture products of vegetable is unique, since the quality is attributed to many of the changes which would be produced in its life cycle. That is, the quality is an important factor of vegetable because it will change inevitably by passage of time. From the point of view of consumer, the good commercial product is defined as an attractive physical appearance and/or its good taste. In this study, the main reason why green bean was rejected is due to the lack of consumer satisfaction on quality. That is, that was the following causes: the size of green bean was too large or small, the color was bruised, and the green bean itself was rotten. If we can reduce the rotten amount, there would be an improvement on the total emission to some extent. In our future plan, we will investigate the effects of cold chain, the buying decision for the eco-friendly vegetable and the promotion of low emission agricultural product system.

5. Conclusion

Due to the quality change of green bean, the total CO_2 emission of product would be different. For this difference, the consumers' behaviors would be changed, and the worse environmental effect would be brought. Hence, in the future work, we will propose a dynamic LCA based on quality change on vegetable.

As described by Annie et al. (2010), a dynamic LCA is proposed to improve the accuracy of LCA by addressing the inconsistency of temporal assessment. For instance, the utilities by which consumers' behaviors would be influenced need to obtain the basic data through the experimental measurement and/or the process investigation. Also, the multi-indexes might be necessary for clarifying the product status, since there are many physical parameters of size, weight, color, edible portion, and nutrients such as sugar, acidity, vitamin and protein content. Based on this study, at the next stage, we will analyze the dynamic life cycle inventory data set of green bean.

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SENSE tool: Easy-to-use web-based tool to calculate food product environmental impact

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ABSTRACT

SENSE project aims to deliver a harmonized evaluation system for the environmental impact assessment of food & drink products. The project evaluates existing relevant environmental impact assessment methodologies, and considers socio-economical, quality and safety aspects, to deliver a new integral system for the environmental and social assessment of agrifood and aquaculture products. The system integrates: (a) (regionalized) data gathering system; (b) matrix of key environmental performance indicators and a (c) methodology for simplified environmental impact assessment. The web-based SENSE tool has been validated in the juice, meat & dairy and aquaculture chains; however, the methodology and its associated software will be modular allowing its implementation in any food product.

The tool is based on the sustainability information collected along the production cycle of any food or drink products. The obtained results are reflected into an Environmental Identification Document (EID). Main results of SENSE will be: (i) Standard key environmental performance indicators (KEPI); (ii) harmonized methodology for environmental impact assessment; (iii) SENSE-tool for environmental data collection; (iv) EID; (v) Certification Scheme Concept (CSC) for sustainability; (vi) Roadmap for policy and governance implementation. SENSE consortium is formed by a multidisciplinary team involving 23 partners from 13 countries made up by a combination of complementary profiles: research organizations, food and drink SMEs, environmental and LCA experts, SMEs for dissemination and communication and European food Associations.

Keywords: Food SME, Online LCA tool, Impact communication, Benchmarking, SENSE Project

1. Introduction

Food production and consumption cause significant strain on the environment as it is estimated that 29% of global emissions of greenhouse gases (GHG) are from agriculture and food production. In the EU, food consumption accounts for 20–30% of various environmental impacts and, in the case of eutrophication, more than 50% (Tukker et al., 2005). In the UK, the food and drink sector is responsible for 14% of industrial energy consumption and 7 Mt of carbon emissions per year; it also uses 10% of all industrial water supplies and produces 10% of the industrial and commercial waste stream (DEFRA, 2006).

The food and drink industry in Europe, of which 99% are small and medium enterprises, is highly fragmented, and food chains are very complex. Hence, to assess the environmental sustainability of a product there is a need for applying integrated, harmonized and scientifically robust methodologies, together with appropriate communication strategies for making environmental sustainability understandable to the market. However, there are difficulties in developing a commonly agreed methodology for environmental impact assessment that still need to be overcome. Challenges are the complexity of food chains, the large number of agents involved, different suitable environmental indicators depending on the business sector, regional differences related to biodiversity among other challenges, including climate change and complexity of the current sustainability assessment tools - high data intensity, costs and expertise required.

Nowadays the calculation of the potential environmental impact of products can lead to great benefits to the industries which, in most cases, can lead to brand differentiation. However, most of the industries in the food sector, especially SMEs, neither have a strong background nor the capability to assess the sustainability of their products.

The European research project SENSE aims to deliver a harmonized system for the environmental impact assessment of food and drink products. The research evaluates existing relevant environmental impact assessment methodologies, and considers socio-economical, quality and safety aspects, to deliver a new integral system that can be linked to monitoring and traceability data. The system will integrate:

(a) (regionalized) data gathering system;
(b) matrix of key environmental performance indicators;
(c) methodology for environmental impact assessment; and
(d) a certification scheme

The methodology will be transferred to food & drink sectors and stakeholders by means of specific communication strategies.

The sustainability information collected along the supply chain of any food stuff and reflected into an Environmental Identification Document (EID) will be accessible by the EID-Communication Platform. This should contribute to making the environmental sustainability part of the usual purchasing behavior of consumers and provide a competitive advantage to those products (and companies) which choose to use the EID. Through a comprehensive environmental communication between the industry and consumers the latter are empowered to choose food products which are environmentally friendly.

2. Methods

2.1. Harmonised methodology for the environmental assessment of food and drink products

A set of consistent environmental impact assessment methods and impact indicators for three food chains (dairy/beef, orange juice and salmon aquaculture) and their supply chains has been selected based on literature reviews. The methodology is based on the key environmental challenges identified for each sector and the related impact categories. For the selection existing methodologies has been reviewed as well as current developments. The ILCD handbook (JRC, 2010) recommends LCIA methods for many impact categories and this has been a starting point for the review. The LCIA methods have also been reviewed on their suitability for the food sector and their practicability to use.

The life cycle assessment methodologies chosen for each impact category are listed in Table 1 along with the corresponding indicators and references. These are the same methods as later recommended by the European Commission on the Product Environmental Footprint (EC, 2013) and in the ENVIFOOD protocol except for water depletion where a revised approach to water footprinting is recommended in the ENVIFOOD protocol (ENVIFOOD, 2012).

Table 1. Life cycle impact assessment methodologies to be used in the SENSE-tool

Impact category	Unit	Selected LCIA method	Reference
Climate change	kg CO2-eq	Bern Model – IPCC	Solomon, 2007
Eutrophication, Terrestrial	molc N-eq	Accumulated Exceedance	Posch et al., 2008
Eutrophication, Freshwater	kg P-eq	EUTREND Model	Goedkoop et al., 2009
Eutrophication, Marine	kg N-eq	EUTREND Model	Goedkoop et al., 2009
Acidification	molc H+-eq	Accumulated Exceedance	Posch et al., 2008
Human toxicity	CTUh	USEtox Model	Rosenbaum et al., 2008
Ecotoxicity	CTUe	USEtox Model	Rosenbaum et al., 2008
Land use	kg $C/m^2/a$	Soil organic matter model	Milà i Canals 2007
Abiotic resource depletion	kg Sb eq	CML 2002	Guinée et al., 2002
Water depletion	m^3 H_2O eq	Ecological scarcity model	Frischknecht et al., 2009

2.2. Key Environmental Performance indicators for food and drink chain

The key environmental performance indicators were proposed as simple parameters to be used in the SENSE tool to calculate the environmental impacts. For the selection of those parameters three LCAs have been done in the beef and dairy, orange juice and aquaculture sectors. The LCA results confirmed the validity of the selected KEPIs taking into account their relevance for the environmental impact, the data availability and the easiness of measurement.

The KEPIs selected for the production of all the food supply chains are shown in Table 2. The selected KEPIs covered 95%, on average, of the environmental impacts of the respective food supply chains (Doubet et al., 2014)

Table 2. Selected key environmental performance indicators for the European food and drink sector.

INPUT	UNIT		DS
Land use	Ha*year	Land occupation for agricultural uses: permanent crops, arable land or grazing.	EcoInvent
Fertilizers	Kg N, P or K/year	Inorganic fertilizer consisting of nitrous compounds such as ammonium nitrate or ammonium sulphate and phosphorous or potassium compounds.	EcoInvent ESU
Organic fertilizer	Kg/year	Fertilizers derived from animal or vegetable matter (e.g. compost, manure	EcoInvent
Pesticides	Kg AI/year	Pesticides are plant protection products. The term "pesticides" covers insecticides, acaricides, herbicides, fungicides, plant growth regulators, rodenticides or biocides. The user has to provide the commercial name for the pesticide (i.e. RoundUp ®) in the free-text box and introduce the amount per hectare used. Once it is defined, an addition table will appear where they have to specify the percentage of active ingredient (AI) (i.e. glyphosate). If the AI is not in the list, generic pesticides could be used, such as, “fungicides” or “herbicides” or “pesticides”. When those AI are used, please introduce the 100% of the content.	EcoInvent
Energy	energy unit kwh, L of diesel, m^3 of natural gas / year	Energy consumption in agriculture systems are mainly related to fuel used during land labors (tractor), energy required for buildings maintenance and greenhouses maintenance, in the fisheries systems to the use of fossil fuel for the fishing vessels and in aquaculture, livestock and food processing systems the energy use is mainly related to the machinery requirements and building general consumption.	ESU
Freshwater use	L or m^3/year	For water requirements the user has to introduce the total water requirements for 1 year. Rain water is not taken into account, only tap-water	EcoInvent
Feeds	Kg/year	Data on feed can be obtained directly from the feed supplier as guest user and should then be added as an incoming product or Data on feed can be selected from a drop down menu, offering different kind of feed ingredients (crop and marine). In the questionnaire, the user should specify the different feed ingredients and add the relative amount by weight.	EcoInvent
Packaging	Kg/year	For the packaging the user should specify the type of final packaging (glass, plastic bottle or so) and the amount used per year. In some cases, intermediate packaging will be relevant too.	EcoInvent
Livestock	nº animals /year	For the livestock, the specific animal has to be selected. Specify the amount produced in one year and the share of the product in turnover (%).	IPCC
OUTPUT			
Wastewater	L or m^3/year	For inland aquaculture systems the user need to specify the amount (l or m3) of wastewater discharges per year. For marine aquaculture systems an average N direct emissions to the marine environmental due to faeces and uneaten feed per kg of fish has been taken into account (Solbakken, et al. 2008).	EcoInvent
Wastes	Kg/year	The user should first choose the waste material (organic waste, plastics, cardboard, glass or other type) and the disposal way (incineration, recycling landfill)	EcoInvent

2.3. Scientific validation of the SENSE tool

The validation of the integrated SENSE tool is based on performing simplified environmental impact assessment oriented to key indicators in the food supply chains representing three food chains (fruit juice, meat and aquaculture fish) in different European regions.

3. Results

Taking into account the methodology selected and the selected KEPIs, a web based tool, the SENSE tool, has been designed and developed with a common server and database allowing an active interaction between users. The developed tool aims to be used by industrial actors without a strong LCA background and to provide easy to be interpreted environmental information.

The tool compiles the information available at different levels in the food chain. The collected data are characterized and evaluated in order to obtain the key indicators associated to the evaluated product. This tool provides a common framework in which users from different stages of the supply chain introduce a simplified set of environmental data and compare respective environmental impacts. As far as the aim of the tool is to provide a tool for the SME's it has been designed as user friendly and very intuitive.

The tool is accessible via internet; therefore it is not necessary to install the program, making its use even simpler. This computer application has been developed using Visual Basic .Net, on Visual Studio 2010. The used database engine is SQL Server 2008 R2, where all the application's information is stored. As far as the application imaging, both design and used pictures, have been done using Photoshop CS 6 y Gimp 2.8.

For the allocation, economic allocation has been selected; however, the tool offers the possibility for system expansion option or to introduce manually the percentage of the economic allocation of different incoming materials, such as packaging or main ingredients.

Moreover, in order to facilitate the data gathering, the tool offers the possibility to send the questionnaires to the main suppliers of the chain. This data is confidential and it will be visible to the user just if the suppliers give the authorization for that.

For the impact characterization, the program sums up the environmental impact of each process involved in the food chain (**Error! Reference source not found.**).

$$T_{x,y} = \sum_{i=x}^{i=z} EI_{i,y} \qquad \text{Eq. 1}$$

Where for any of selected environmental impact, such as climate change or water depletion (Table 1):

- $T_{x,y}$ is the summation of the environmental impact of all inputs *j* of the product *y* from the process *x* to the last process *z* (where final product are generated)
- $EI_{i,y}$ is the environmental impact of the process *i* in the product *y*

For the impact assessment of each process, the proportional impact of each input into the final product has been added according to equation 2 (**Error! Reference source not found.**).

$$EI_{x,y} = \sum_{\forall j} I_{j,x,y} \qquad \text{Eq. 2}$$

Where:

- $EI_{x,y}$ is the environmental impact of the process *x* in the product *y*
- $I_{j,x,y}$ is the partial environmental impact of the input *j* of the process *x* in the product *y*

Finally, to calculate the proportional environmental impact of each input in the final product, the conversion factor of each input to each environmental impact has been multiplied for the amount of the input used in each process. Then a factor is applied to calculate the weighting of this input into the target product. This factor is calculated taking into account the share of product turnover of the processes involved and the ratio of the product of each process used in the next process (Eq. 3).

$$I_{j,x,y} = CF_j \times M_{j,x} \times \prod_{i=x}^{i=z} (S_{y,i} \times R_{y,i}) \quad \text{Eq. 3}$$

Where:

- CF_j is the environmental characterization factor for the selected impact category of the input *j*
- $M_{j,x}$ is the quantity of the input *j* in the process *x*
- $S_{y,i}$ is the share of product turnover in the process *i* for the product *y*
- $R_{y,i}$ is the ratio of the product flow between the process *i* and *i+1* for the product *y*

For example, for the dairy chain described in
Figure 1, in order to calculate the impact of the farm process on the final product (i.e. sour cream) accordingly to the formulas described above, the procedure will be the following:

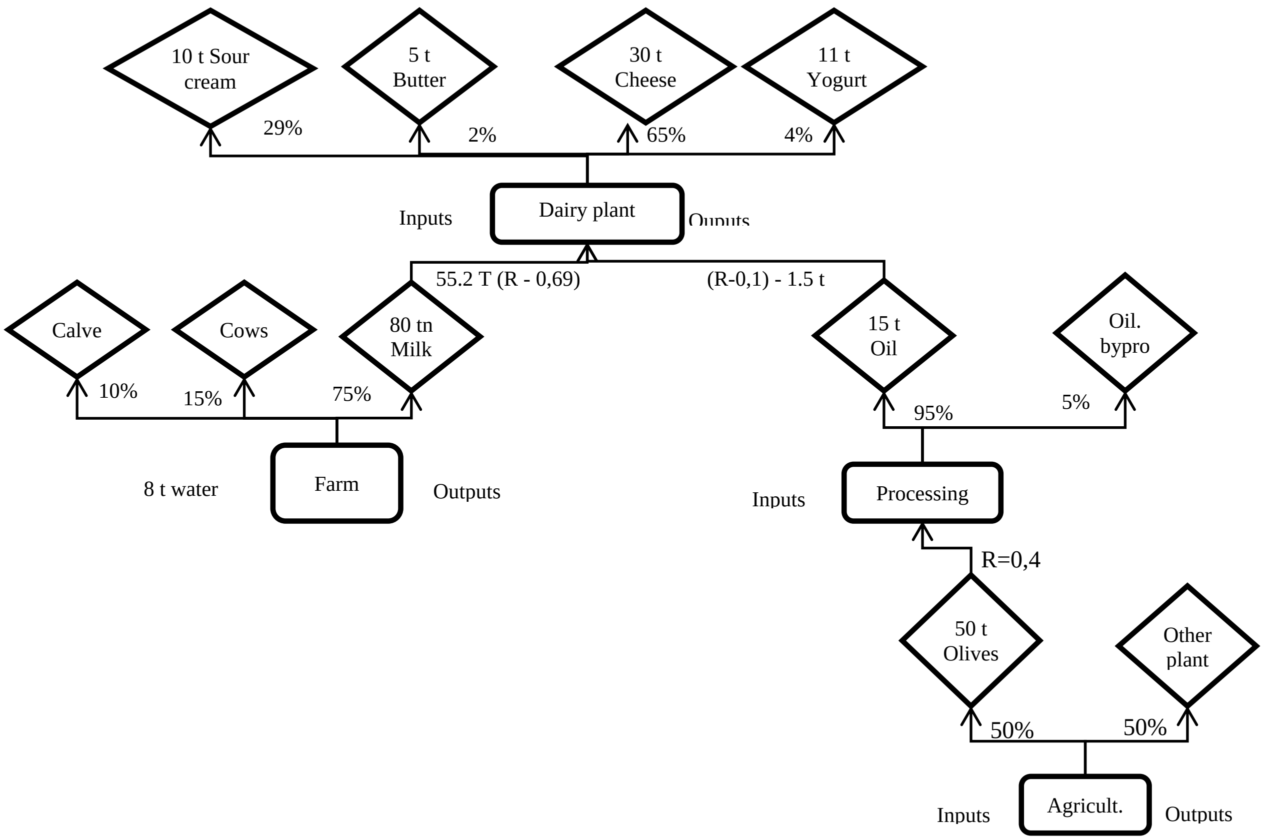

Figure 1. A hypothetical dairy plant which produces sour cream, butter, cheese and yogurt. Main ingredients are raw milk from a farm and oil. Percentages in the lines represent the share of product turnover and the Ratios (R) represent the ratio of product taken from the previous stage.

Taking into account that 8 tn of water enters the farm and that the characterization factor for the climate change of water is 0,0003 kg CO2 eq/kg water, the proportional climate change potential of the water consumption regarding the sour cream yearly production will be the following:

$$I_{j,x,y} = 0{,}0003 \times (8 \times 1000) \times 75\% \; x \; 0{,}69 \; x \; 29\% = 0{,}36 \; kg \; CO2 \; eq$$

Hence, adding all the proportional climate change impact of all the inputs to the farm stages, the user could differentiate the impact of that specific process. Thus, adding the processes involved in the whole chain, the complete impact will be obtained.

The SENSE tool application calculates the environmental impacts that are related to the previously described impact categories. The impact characterization can be shown for the production of one year, or for one unit e.g. kg product as defined in the user profile.

The results are presented in the following ways:

- Environmental impact per process and year. Those results are shown in a bar chart and show the impact generated for the selected environmental indicator and process. A table with the impact value is also shown under the graphic. (Figure 2)

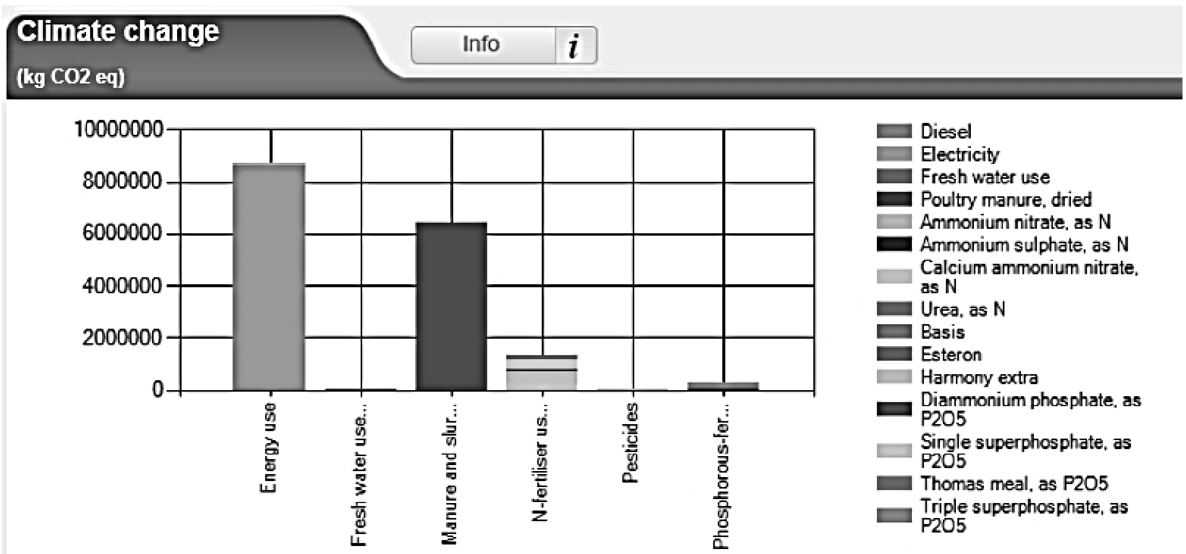

Figure 2. Captured figure of the sense-tool results for the climate change potential characterization results for the farm stage of the dairy production chain.

- Complete impact analysis: For each impact category a pie graph is shown with the contribution of each process to the total impact. An histogram is also shown with the summary of all the impact and processes (Figure 3)

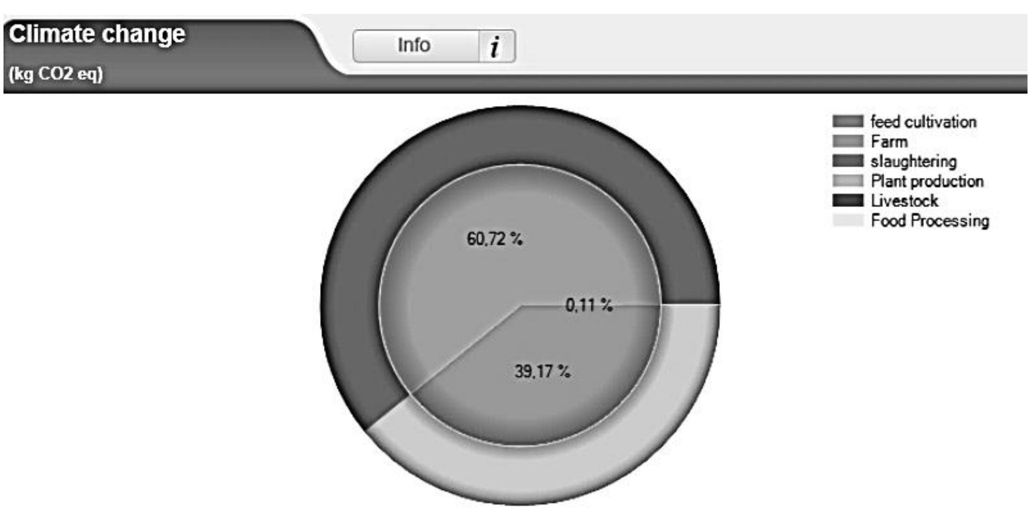

Figure 3. Captured figure of the sense-tool results for climate change potential characterization for a dairy production chain expressed in pie chart.

The contribution (%) of each process to the final impact is shown.

- Compare products. It is possible to compare product by process or by impact. When comparing the environmental impact by process, the weight of the different processes on the final impact of each product will be shown for each impact category. When comparing by impact, a complete graph will be shown comparing the final impact of each product.
- Evolution of the product impact. A line chart is shown with the evolution of the environmental impacts of the product along the years. With this data the tendency can be assessed.
- Product benchmarking: In the future this option will allow the user to benchmark its products internally (comparison between the same products in different years) and externally (with other similar products). When selecting benchmarking a spider graph will be shown with the deviation of the actual product impact assessment from the average value for that product. For the moment, there is no enough data for the external benchmarking; however the tool is ready for the future improvements.

Coupled with those graphics, there is a possibility to extract the Environmental Identification Document (EID). In this document a summary of main environmental results is described.

4. Discussion

4.1. Validation of the tool

The tool has been technically validated in two case studies by comparing the outcome of the SENSE tool with calculations performed by SimaPro commercial software Simapro. The functionality of the SENSE tool was tested when entering data for the beef-and-dairy, orange juice. Acuaculture chain has been also validated with GaBi software, but the results of this comparison are out of the scope of this publication. The validation is based on using only pre-selected input data (e.g. energy use, material use, etc.) which have been defined in the project as key performance indicators (KEPIs). The KEPIs were chosen based on their contribution to the key environmental impacts of the food supply chains studied namely, beef and dairy, orange juice and aquaculture fish (Doublet et al., 2014). This iterative process was important to ensure that the developed SENSE tool would be fully functional and validated before it was delivered for implementing and testing by SMEs.

The relative percentage difference between the environmental impacts of the SENSE tool and the LCA on Romanian beef and dairy products (Doublet et al. 2013a) is highly dependent on the impact category. Results for climate change, human toxicity cancer and non-cancer effects, ecotoxicity, freshwater and land use have a difference smaller than 10 %. However, differences in the modelling of the emissions due to the land use and the application of, manure as well as the additional data taken into account in the complete LCA for the pesticides can explain the large deviation in the results of the acidification, eutrophication terrestrial and marine.

For the orange juice supply chain, the difference between the environmental impacts calculated in the SENSE tool and the LCA is below 10 % for climate change, human toxicity, acidification, eutrophication terrestrial, eutrophication marine, abiotic resource depletion and water depletion.

4.2. Allocation procedures

Since the aim of the project is to obtain a simplified environmental analysis of the food and drink products, some limitations have been identified. The method used when distributing the environmental burden between the main product and its by-product can have a significant impact on the final results of a LCA (Svanes et al., 2011). Although it may be controversial, economic allocation is chosen as the default allocation approach in the SENSE tool.

The allocation procedures applied in the LCA on beef and dairy products in Romania are beyond the common knowledge of SMEs. The allocation procedures recommended by the international dairy federation to allocate the environmental impacts of beef and milk production at farm as well as the allocation matrix to distribute the environmental impacts of the individual dairy products are too complex and time-consuming for somebody not familiar with the field of life cycle assessment. Witczak et al. (2014) conclude that SMEs do not have time to collect and evaluate data and expect quick results based on a small amount of data. ISO recommends avoiding allocation by expanding the system but this is out of the scope of an internet tool to be used by SMEs. Allocation cannot be avoided and allocation rules should be made as simple as possible. The easiest allocation approaches are mass and economic allocation.

The results for single dairy products are quite sensitive to the allocation approach chosen (Feitz et al. 2007). Physico-chemical allocation, mass allocation, protein allocation and economic allocation were used to assess the environmental impacts of individual dairy products. Mass allocation may be discredited in the dairy production chain since it results in considerable deviations from physicochemical allocation. Economic allocation introduces similar order of magnitude sized variations. Feitz et al. (2007) suggested using economic allocation for inter-industry sectorial flows. Kim et al. (2013) allocated the incoming raw milk to the individual dairy products on a milk solids basis. Energy and resource use were allocated based on an economic allocation.

The allocation of environmental impacts to by-products is also an issue for the slaughtering process in the beef chain. Due to lack of comprehensive global data, Opio et al. (2013) could not perform an allocation to slaughter by-products. Cederberg et al. (2009) explained that no greenhouse gas emissions were allocated to the meat production by-products. In our case study, this approach was followed and all environmental impacts are allocated to the beef.

In the aquaculture chain, the use of economic allocation has been criticized as it does not reflect the biophysical properties of the production system and is sensitive to changes in market prices (Pelletier & Tyedmers, 2011; Svanes et al., 2011; Ytrestøyl et al., 2011). Mass allocation methods have been applied in studies on feed and aquaculture as well as fisheries (Boissy et al., 2011) while others have used gross nutritional energy (Pelletier et al., 2009) or economic allocation (Ellingsen et al., 2009). Winther et al. (2009) justified the use of mass allocation for salmon after evaluating both economic allocation and gross nutritional energy. In mass allocation, the environmental cost associated with the by-products is the same as for the products for human consumption. Using mass allocation in LCA is beneficial for producers of products for human consumption if they can recycle their by-products into other production systems. Therefore, mass allocation creates a positive incentive for full utilization of by-products compared to economic allocation, where by-products of insignificant value otherwise carry a zero environmental burden. However, the use of by-products from environmentally costly productions such as livestock production or demersal fish trimmings in salmon feed production contribute substantially to the outcome of an LCA analysis in terms of energy use and CO2 emissions (Pelletier et al., 2009; Ytrestøyl et al., 2011). Currently about a quarter of the fish meal produced comes from by-products from fish processing for human consumption (i.e., by-products from fish filleting plants). In the case study, the economic allocation was used in the LCA on aquaculture. It gives a higher burden on the main product than if mass allocation would have been used. At the aquaculture farm 10 % of the biomass at the farm is guts which are given away for free. The by-product, guts, therefore has zero environmental loads. If mass allocation would have been used the impacts of the salmon product would be reduced by 10 %.

The recommendation regarding economic allocation rules for the SENSE tool may be the simplest approach for SMEs. However, since the SENSE tool offer the possibility to implement different allocation factors for the incoming product; this is a good approach that could be used if SMEs are willing to invest more time to obtain a more scientific environmental assessment.

4.3. Usefulness in SMEs

Additional case studies where the SENSE tool is tested by users are currently ongoing in the project in at least 30 companies and their supply chains. First impressions with the SMEs state that the companies are quite reluctant to implement the SENSE-tool into their company mainly due to lack of resources (time or people). However, after this first obstacle, those companies which are taking part of this validation find it very useful. Main benefits of the tool for those companies are i) the possibility to identify the hot-spot of their processes and ii) the benchmarking possibility (not implemented yet).

5. Conclusion

In conclusion the SENSE tool has been designed to be suitable for the food and drink SMEs. However, it is important to remark that the main aim is to obtain a simplified tool, and thus it won’t be an alternative for the complete LCA studies.

For future developing two mains aspects will be considered. First the Certification Scheme Concept voluntary system (CSC) for use of the EID in food and drink products will be developed. This Scheme will be based on the different voluntary systems (ISO, EMAS, others). The CSC will define the rules for development of its own regulations for the certification process, the definition of the requirements (independence, transparency, etc.) and the protocol of the regulatory/certification system which will be in charge of the approval and updating of the EID. The second important aspect to develop in the future is the communication systems which will be differentiate between business-to-business (B2B) and business-to-consumer (B2C) communication levels.

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Modeling pesticides emissions for Grapevine Life Cycle Assessment: adaptation of Pest-LCI model to viticulture

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ABSTRACT

This paper presents a tailored version of PestLCI 2.0, the most advanced life cycle inventory model for quantification of organic pesticide emissions from arable land, customized to appropriately account for viticulture specificities affecting pesticides emissions from vineyards. PestLCI 2.0 customization is further supported by the calculation of USEtox™ freshwater ecotoxicity characterization factors for active ingredients relevant in viticulture. Case studies on two different vineyard management systems illustrate PestLCI 2.0 model application. The customization of the PestLCI 2.0 model includes addition of 29 active substances, 9 application techniques, interception by a dual canopy (vine /grass cover), new soil and climate databases and further account of multiple vinerow treatment. Four substances dominate the overall toxicity profiles. Comparing the results obtained with PestLCI 2.0 with existing static emission quantification approaches results reveals that PestLCI 2.0 yields considerable lower emission loads and consequently, lower toxicity impact burdens. The issue of accounting for inorganic substances is discussed.

Keywords: plant protection product, LCA, vineyard, modelling, emissions, fate

1. Introduction

Wine production and hence viticulture are an important economic and cultural sector in main parts of the world including Europe. Viticulture is also a major pesticide consuming sector with app. 20 kg formulated pesticides/ha/year (of which app. 15 kg/ha are elemental sulfur) within the European Union (Endure 2010).

Pesticides emissions are therefore a crucial topic to be addressed when performing wine and/or grape production related life cycle assessments (LCAs). Due to the lack of specific inventory models suited for pesticide emission quantification, most of the published wine LCA studies either neglect pesticide emissions (Point et al. 2012; Pattara et al. 2012; Bosco et al. 2011; Ardente et al. 2006), simply assume that 100% is emitted to the soil (in accordance with the default Ecoinvent approach applied for pesticides (Nemecek and Schnetzer 2011)) or propose a fixed partition as (Neto et al. 2012) with 75 % of pesticides emitted to soil and 25% to the air.

PestLCI (Dijkman et al. 2012; Birkved and Hauschild 2006) is currently the most advanced LCI model for pesticides emissions quantification from arable land (van Zelm et al. 2014). Despite the fact that the model has been applied in wine/grape production LCAs (Villanueva-Rey et al. 2014; Vázquez-Rowe et al. 2012), PestLCI 2.0 still does not take into account certain viticulture specificities such as double cropping systems, vertical application techniques etc. which differentiate viticulture from other crops and influence the emission patterns of pesticides. Characterization of the impacts from pesticide related emissions specific to viticulture and fruits or vegetable crop production furthermore requires characterization factors for these chemical compounds.

This paper presents a PestLCI 2.0 version customized to appropriately account for viticulture specificities affecting the emission quantities and patterns of pesticides from vineyards. The application of the customized inventory model is illustrated by calculation of potential ecotoxicity impacts through combination of emission quantities and freshwater ecotoxicity characterization factors calculated using the USEtox™ characterization model. The combination of the PestLCI 2.0 and USEtox™ models is illustrated through two case studies on vineyard technical management routes[1] (Renaud-Gentié et al. 2014; Renaud-Gentié et al. 2013).

[1] technical management routes (TMRs): logical successions of technical options designed by the farmers (Renaud-Gentié et al. 2014))

2. Methods

2.1. Pest LCI

PestLCI is a dedicated inventory model intended to provide emission quantities for use in relation to the characterization of chemically induced impacts caused by emissions of pesticides from arable land/technosphere to the environment/ecosphere. The most recent version of the model, version 2.0, is described in Dijkman et al. (2012). This version of the model covers app. 90 active ingredients of various types of pesticides, 25 European climate profiles and 7 European soil profiles.

Despite the rather extensive coverage in terms of pesticides, climates and soils, the most recent model version was developed with conventional agriculture in mind and viticulture was hence not addressed specifically. In order to improve the viticulture specificity of the model, PestLCI version 2.0 was updated with 28 pesticides frequently used in European viticulture, 5 French soil profiles, 22 French temperate maritime climate profiles, 34 crop stages and cover crop combinations + 13 direct spraying situations as well as 9 pesticide application techniques typically applied in European viticulture.

2.2. Identification of specific needs for pesticides emissions modeling in viticulture

2.2.1. Pest and weed management in viticulture

The main pests damaging the vine canopy are primarily the fungi downy mildew *(Plasmopara Viticola)* and powdery mildew *(Uncinula Necator*), which necessitate fungicide treatments. The other fungi and the main insect pests (moths, leafhoppers and phytophagous mites) are not systematically treated. Vineyard management includes also weed control, since weed presence can affect vine growth by competition for water and nutrients. Most of these pests require specific pesticide active ingredient substances (ai.s). The risk of resistance acquisition by the pests implies frequent change of ai.s, in conventional[2] viticulture especially for ai.s presenting a single-site action. Vineyard treatment programs therefore usually involve a variety of pesticide ai.s. This is however not the case is in organic viticulture, where the fungicides used (primarily copper and/or sulfur based fungicides) have multi-site action, while weeds in organic viticulture are mechanically controlled.

2.2.2. Active substances

Vineyard management involves, in French context, on average 14 to 16 pesticide applications per year at full recommended application rates (Mézière et al. 2009; Ambiaud 2012b) with a high interregional and inter-annual variability mainly related to climatic conditions and winegrowers practices. Organic and inorganic or partially inorganic pesticides ai.s are used by viticulture.

Conventional pest management in viticulture relies on generic farming organic pesticides ai.s, but also on more crop specific organic pesticide ai.s typically used on vegetables or in orchards. All these crop specific active substances were not included in PestLCI 2.0 version presented by (Dijkman et al. 2012). The substances used in the cases studied in the project at hand were listed. Compilation of data on the properties of these substances used in viticulture, however missing in the PestLCI 2.0, was conducted applying specialized chemical databases: "e-phy" (MAAF and ONPV 2013) for correspondence between commercial name and active substance, "PPDB" (University-of-Hertfordshire 2013), "Toxnet" (US-National-Library-of-Medicine 2013) as well as "Chemspider" (Royal-Society-of-Chemistry 2013) for main chemical and physical characteristics. Data gaps were compensated for applying the Quantitative Structure-Activity Relationships included in the EPI Suite™ (US-Environmental-protection-Agency 2012).

However, conventional and organic viticultures also apply inorganic sulfur (S8) and copper-based fungicide ai.s. to manage powdery mildew (mainly sulfur) and downy mildew (mainly copper). They also use other inorganic ai.s like Ammonium thiocyanate (herbicide) or partially inorganic like Fosetyl-Al (fungicide). However, PestLCI 2.0 is designed only for modelling of organic pesticide a.i. emissions. For this reason, inorganic pesticides were not included in this study.

[2] « Conventional » is used in this paper to designate non-organic plant protection practices

Some pesticide ai.s applied in the studied vineyards, especially organic vineyards are derived from micro-organisms like Spinosyn, or plants like Pyrethrum. The physical-chemical and fate properties of these pesticide ai.s were found in Bio-Pesticide DataBase (BPDB) (University-of-Hertfordshire 2012) and some of the previously cited databases. A second fraction of “others pesticides and pesticide ingredients” are substances like algae extracts and pesticide formulation additives which are not considered in this study, due to lack of information about the nature and quantity of these substances. Formulation additives may however despite of their exclusion of this study, contribute considerable to toxicity of the pesticide formulation (Brausch and Smith 2007).

2.2.3. Spraying equipment

Vertical shoot positioning is the most frequent training system[3] for vineyards in France and is found in many other wine producing countries. The case studies assessed in the paper at hand only cover vertical shoot positioning trained vineyards.

The types of spraying equipment involved in viticulture are multiple, which makes the task of modeling their individual characteristics a challenge. Herbicides are most often applied using specific sheltered booms to avoid herbicide drift and hence deposition on vine leaves (which would damage/kill the wines). We chose to model these application techniques as “soil incorporation” in Pest LCI since they induce very low drift. The sprayers designed for canopy and grapes spraying can use different modes of droplets production:

- *non air-assisted spray*, the droplets are formed by the acceleration of the liquid under pressure in the nozzle, and applied to the leaves by the pressure,
- *airblast*, the droplets are formed through nozzles, and are then subsequently transported to the leaves by an air flux which further shakes the wine leaves, thereby facilitating the penetration of the substance into the canopy,
- *pneumatic* where air accelerated to app. 300km/h meets the liquid containing the pesticide formulation and fragments the solution into very fine droplets and conveys it to the leaves.

Different shapes of the ventilators and of the sprayers themselves lead to different patterns in terms of spraying quality and drift generation. These different shapes also permit either a direct spraying of each face of the canopy or a spraying from the top of the rows.

A last type of sprayer was designed for drift reduction: the tunnel sprayer where specific panels prevent from drift and collect the non-intercepted spray mixture and recycle it for re-use.

PestLCI 2.0 takes into account the type of sprayer in order to quantify the drift calculations through drift curves. Sprayers representing the above presented application techniques were however not available in PestLCI 2.0. In the present customization, 9 sprayers with different modes of action were included. For the customization and the viticulture specific application techniques we used data from (Codis et al. 2011) who conducted drift tests according to International Organization for Standardization (ISO) protocol (ISO 2005), on different vineyard spraying equipment types in France. The sprayers successfully tested by (Codis et al. 2011) plus a tunnel sprayer tested by (Ganzelmeier 2000) were included in the viticulture customized PestLCI 2.0 version. Defining within this list of the 9 sprayers, the ones that are the more similar to the ones we have in our case studies was done through discussion with the author S. Codis (Codis 2014).

Since the aforementioned drift tests used to derive spray drift curves were conducted on vines with one leaf area, PestLCI 2.0 was adapted in order to correct for spraying on vines with different leaf areas. Modelling of custom spray techniques covering various adaptations of existing spraying equipment is considered beyond the scope of this paper.

According to the type of sprayer, the winegrower can choose to spray 1 to 4 rows of vines simultaneously. The number of rows treated plays a role in wind drift calculation in PestLCI 2.0. It has been taken into account using the parameter” nozzle distance on the model”, the distance entered being the actual width (in meter) treated at the same time.

[3] Training system : type of trellis and shoot positioning resulting to a given shape of the vine canopy and position of grapes.

2.2.4. Accounting for primary distribution in double cropping system

The primary distribution process is defined in PestLCI by 3 factors: wind drift (f_d), pesticide deposition on soil (f_s) and pesticide deposition on leaves (f_l) (Birkved and Hauschild 2006). The two latter are based on (Linders et al. 2000) interception factors for single crops at different growing stages.

Concerning interception by vine canopy, PestLCI 2.0 includes interception values for vine at four different development stages I, II, III, and IV based on (Linders et al. 2000). A stage 0 has been added to take into account situations of leafless vines. The interception factor has been estimated from the orchard dormancy stage interception factor (0.2) given by (Linders et al. 2000) with a division by 2 due to the difference of perennial parts importance between fruit trees and vines. The value of f_l for vine 0 was therefore set to 0.1.

On-field measurements of spraying mixture deposition and losses on vineyards were made by Sinfort (Sinfort 2014) (Sinfort et al. 2009) and on artificial vineyard by (Codis 2014). Distribution ratios of spray mixture between vine canopy, soil and air at 2.5 m above the soil were obtained by these authors in vineyard conditions similar to the ones we study (rows width, types of sprayers). The fractions going to air during spraying measured by (Codis 2014) and (Sinfort et al. 2009) were considered for introduction in PestLCI 2.0 as being i)partly conveyed by wind drift out of the parcel (i.e. advective transport), and ii)partly falling back on vegetation and bare soil of the parcel (i.e. sedimentation). This is because no quantification of direct volatilization is possible (Jensen and Olesen 2014) due to the complexity of drivers combination (properties of the spray liquid, drops size and drops surrounding conditions)(Gil et al. 2007) and the lack of available data for some of them. As PestLCI 2.0 calculates the quantity of drifted pesticide on the basis of the dose applied and before calculating leaf interception, we decided to apply a drift quantity correction ratio based on the pesticide fraction going to air. Full vegetation (stage III) was given the 1:1 ratio because sprayers drift curves were established on that stage.

For accounting of direct spraying with hand hose in early stages or young vineyards, and direct spraying on grapes, 13 specific interception factors were defined through expertise.

Cover cropping on vineyard soil is a developing management scheme with nearly half of French vineyard temporarily or permanently covered (Ambiaud 2012a). It offers multiple advantages, provided the competition for water and nitrogen isn't excessive for the vineyard (Celette et al. 2009). This second canopy under the vineyard (e.g. spontaneous species, oats, clover or fescue) contributes to pesticide interception (primary distribution) and fate (secondary distribution). The interception by the cover crop varies according to the width of the cover crop strips estimated in percentage of the width of the vine inter-row, and according to cover crop canopy density.

In the case of mixed cropping (vine + cover crop), a complementary interception factor (*f*) needs to be added for cover crop. The structure of PestLCI 2.0 being fixed with 3 *f* entries, a combined *f* has been calculated from f_{vine} and $f_{covercrop}$. Based on calculation routines, 34 combined interception factors are now available in the customized PestLCI 2.0 model covering the most current situations spanning possible combinations of vine development stages, cover-crop strip width and cover crop canopy density. Examples of such combinations are given in Table 1.

Table 1: Examples of combined interception factors for vine/cover crop mixed cover (*f* = interception factor)

Stage	**cover density**	**% of soil surface covered by grass**	f_{vine}	$f_{covercrop}$	**% spray lost in air**	**% intercepted by vegetal soil cover *(calculation)***	f_{global}
0	none	0	*0.1*	0.3	*30%*	0%	0.10
II	weak (30%)	100%	0.5	0.3	*30%*	6%	0.56
II	high (70%)	80%	0.5	0.7	*30%*	11%	0.61
III	average (50%)	100%	0.65	0.5	*25%*	5%	0.70

2.2.5. Databases

Site specific climatic profiles appropriately addressing the case study areas were introduced in PestLCI 2.0: two sets of 30 years average 1971-2000 and 1981-2010 for the Beaucouzé Station, and for the five stations of the Middle Loire Valley, being the closest to the observed parcels, data for 3 years of production i.e. October n - September n+1, for 2009-2010 to 2011-2012, a set of averaged months over the 3 years is also available. Climatic data were provided by Météo France. The soils of the modelled parcels were characterized through measured data and observations, in accordance with the PestLCI 2.0 data requirements, and entered in PestLCI 2.0.

2.3. Case study

2.3.1. Choice and main characteristics

Two quite different technical management routes (TMRs) of *Chenin Blanc* cultivar in the middle Loire Valley (France), were observed during the production year 2010-2011. The prevailing Loire Valley climate is temperate maritime according to the classification used in PestLCI 2.0. This study is focused on vineyard TMRs designed for Protected Denomination of Origin (PDO) dry wine production from this cultivar. The case studies were chosen within the 5 types of vineyard TMRs resulting from a detailed survey and analysis of existing vineyard management practices in the area (Renaud-Gentié et al. 2014): 1 "systematic synthetic chemical use and limited handwork",2 "moderate chemical use", 3 "minimum synthetic treatments and interventions",4 "moderate organic", and 5 "intensive organic"(organic with many interventions and treatments).

The two cases presented here: TMRs 1 and 3 are real situations representing respectively type 1 and 3. Organic TMRs are not studied here because their treatment programs include mainly or exclusively inorganic pesticide ai.s. No surface water body lies at less than 100m of the parcels. The 2 plots are not drained nor irrigated.

In comparison to the 30 year average (1981-2010) of the area, the climate of the focus year is: i) a little warmer (0.2°C above the annual average) with a warmer spring but a cooler summer, ii) much drier especially during the growth season of the vine.

TMR 1's plot climate is described by Blaison-Gohier weather station, is on the soil "UTB131" according to the regional cartography of vineyard soils, with 5% slope. TMR 3's plot climate is described by Fontaine-Guérin station. Its slope is 3%, and soil is "UTB35". Field width and length were put at 100m for the two plots. Vineyard pest management programs, spraying equipment and pesticide ai.s used by the growers, as well as canopy development stage and grass cover density and extent are described in table 2.

Table 2: Characteristics of pesticide ai.s applications for 2011 on TMRs 1 and 3. *In grey: the inorganic and partially inorganic pesticide ai.s that were not included in this study.*: new entries in PestLCI 2.0. ▪ needed a new CF in USETox^{TM}*

	pesticide ai.s	Application rate	Crop type + development stage	Month of application	Application method	width treated at a time
TMR1	*Amitrole	0.79	Grass I - all phases	april	sheltered boom	1.85
	Aclonifen	0.31	Grass I - all phases	april	sheltered boom	1.85
	Sulfur	5.89	*Vines II - h80% grass	may	tunnel sprayer	1.85
	Folpet	0.74	*Vines II - h80% grass	may	tunnel sprayer	1.85
	Fosetyl-Aluminium	1.47	*Vines II - h80% grass	may	tunnel sprayer	1.85
	*Fluopicolide	0.12	*Vines II - h80% grass	may	airblast sprayer	7.4
	Fosetyl-Aluminium	1.75	*Vines II - h80% grass	may	airblast sprayer	7.4
	*▪Proquinazid Technique	0.05	*Vines II - h80% grass	may	airblast sprayer	7.4
	*▪Tétraconazole	0.03	*Vines III - a80% grass	june	airblast sprayer	7.4
	*Indoxacarbe	0.04	*Vines III - a80% grass	june	airblast sprayer	7.4
	copper oxychloride	0.73	*Vines III - a80% grass	july	airblast sprayer	7.4
	copper sulfate	0.18	*Vines III - a80% grass	july	airblast sprayer	7.4
	*▪Cymoxanil	0.12	*Vines III - a80% grass	july	airblast sprayer	7.4
	Mancozèbe	0.40	*Vines III - a80% grass	july	airblast sprayer	7.4
TMR3	Glyphosate	0.54	Grass I - all phases	march	sheltered boom	1.95
	*Amitrole	0.92	Grass I - all phases	march	sheltered boom	1.95
	ammonium thiocyanate	0.86	Grass I - all phases	march	sheltered boom	1.95
	*▪Flazasulfuron	0.02	Grass I - all phases	march	sheltered boom	1.95
	Glyphosate	0.09	Grass I - all phases	may	sheltered boom	1.95
	▪Trifloxystrobin	0.06	*Vines II - a50% grass	may	pneumatic sprayer side by side	7.8
	▪Trifloxystrobin	0.06	*Vines III - a50% grass	june	pneumatic sprayer side by side	7.8
	Diméthomorph	0.18	*Vines III - a50% grass	june	pneumatic sprayer side by side	7.8
	Mancozèbe	1.20	*Vines III - a50% grass	june	pneumatic sprayer side by side	7.8
	*Difénoconazole	0.03	*Vines III - a50% grass	july	pneumatic sprayer side by side	7.8
	*▪Meptyldinocap	0.21	*Vines III - a50% grass	july	pneumatic sprayer side by side	7.8

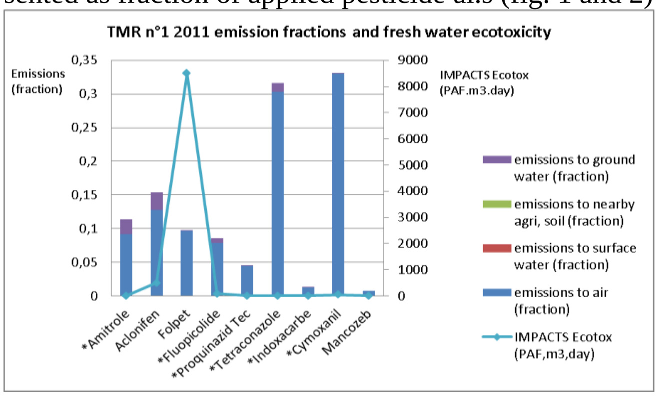
2.3.2. Calculating characterization factors in USEtox™

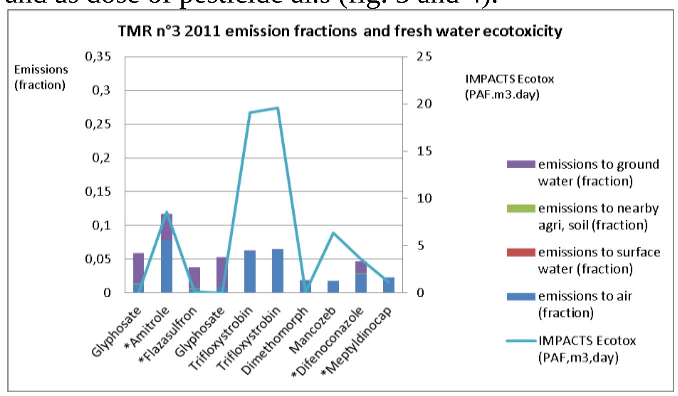
Characterization factors (CF) are needed to quantify the potential environmental impact from emissions of the life cycles of products and systems in LCA. CFs are substance and compartment specific and CFs for each of the emitted ai.s of the case study for compartments air, surface water and agricultural soil[4] are thus needed. We choose USEtox™ as characterization model since it was developed as a scientific consensus model, to represent the best application practice for characterization of toxic impacts of chemicals in LCA (Hauschild et al. 2008) and since its database (v. 1.01) covers ~2500 chemicals with calculated characterization factors for freshwater ecotoxicity (Rosenbaum et al. 2008). Some of the ai.s were not covered by the database and we therefore applied the USEtox™ model to calculate them. USEtox™ requires a number of chemical and physical properties in addition to toxicity data. Primary and secondary sources for these data are available upon demand to the authors.

3. Results

3.1. Emissions and Freshwater ecotoxicity on the case study

Pesticide ai.s emissions were calculated by PestLCI 2.0 for each organic substance application. They are presented as fraction of applied pesticide ai.s (fig. 1 and 2) and as dose of pesticide ai.s (fig. 3 and 4).





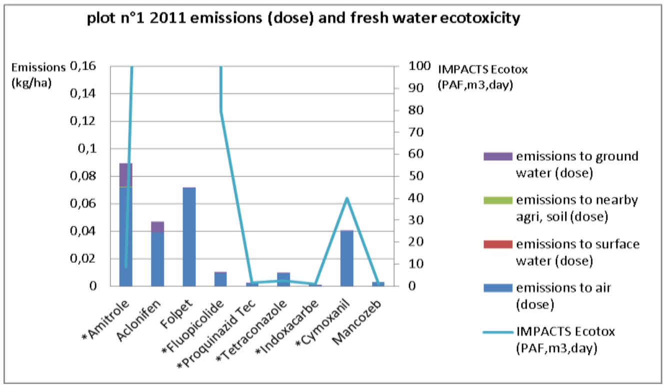
Figures 1 and 2: fraction of applied pesticide ai.s emitted in the 4 compartments: fresh and ground water, nearby agricultural soil and air (left axis) and the resulting freshwater ecotoxicity (right axis, scales adapted to results)

The total emission fractions do not exceed 0.35, and are lower than 0.15 for most of the pesticides. The emissions loads are dominated by air emissions in figure 1, followed by ground water emissions. In figure 2, groundwater emissions dominate, for the 4 herbicides. Emissions to nearby agricultural soils are negligible, and thus don't appear on the charts. The absence or quasi-absence of freshwater emissions are due to the absence of water body around the parcels.

The higher total emission fractions relate primarily to Tetraconazole (fungicide of the Triazole group), Cymoxanil (fungicide of Cyanoacetamide Oxime group) and Mefenoxam (fungicide of the Phenylamide group). These fungicides are emission load wise followed by two herbicides: Aclonifen (Diphenyl ether group) and Amitrole (Triazole group) and another fungicide: Folpet (Phtalimides group).

In some cases, the same substance is applied under two different conditions. Amitrole, for example, is applied with the same type of boom (herbicide sheltered boom) and on the same canopy (grass). However, the emissions of this compound vary according to the conditions of application (month of application and soil vary). An application of Amitrole in April (figure 1) compared to an application of Amitrole in March (figure 2) shows lower emissions to air for the early application, while emissions to groundwater are higher for the latest application.

[4] USEtox contains no ground water compartment. Ecotoxicological impacts in freshwater from chemical emissions to groundwater are considered negligible and thus not further considered in this study.



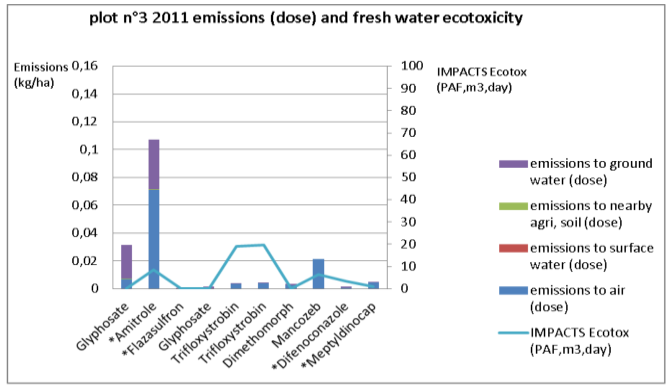


Figure 3 and 4: Quantity of applied pesticide ai.s emitted in the 4 compartments and freshwater ecotoxicity (Impacts Ecotox (same values as fig 1 and 2) scale was fixed here to a maximum of 100 to see smaller differences)

When expressing the emissions in kg/ha, see figures 3 and 4, the quantity emitted to the ecosphere per application is not higher than 0.12 kg/ha. Amitrole dominates the emissions air in both scenarios (fig. 3 and 4), followed by Folpet and Aclonifen.

Freshwater ecotoxicity (FwEtox) was calculated applying USEtox™ characterization factors. Figures 1 and 2 reveal considerable differences for the different application times. In figure 1, FwEtox is dominated by Folpet (8000 PAF·m^3·day), Aclonifen (500 PAF·m^3·day), Fluopicolide (80 PAF·m^3·day) and Cymoxanil (40 PAF·m^3·day). The high impact from Folpet emissions is primarily caused by emissions to air and can be explained from a combination of the substance's relatively low vapor pressure (meaning that a substantial amount of emissions of Folpet to air ends up in the freshwater compartment) and its relatively high toxicity to freshwater organisms. TMR3 shows a much lower total FwEcotox (58.5 PAF·m^3·day) than TMR1 (9132 PAF·m^3·day).

3.2. Comparison with static emission quantification approaches

The Ecoinvent approach applied for pesticides is to consider that 100% is emitted to the soil (Nemecek and Schnetzer 2011), (Neto et al. 2012) in their LCA of Portuguese wine Vinho Verde propose a static partition as with 75 % of pesticides emitted to soil and 25% to the air. Table 2 compares results between the 3 approaches.

Table 3: Comparison of pesticides emissions calculated by PestLCI 2.0, Ecoinvent and Neto et al. (2012) approaches and FwEtox calculated with USETοxTM characterization factors

Emissions	average fraction emitted	standard deviation on fractions
PestLCI 2.0-emissions to air	7.39E-02	9.30E-02
PestLCI 2.0-emissions to surface water	0	0
PestLCI 2.0-emissions to nearby agri. soil	8.06E-05	1.19E-04
PestLCI 2.0-emissions to ground water	1.32E-02	1.68E-02
PestLCI 2.0-total emissions	8.72E-02	9.21E-02
100% emitted to soil (Ecoinvent)	1	0
75% emitted to soil (Neto etal 2012)	0.75E-01	0
25% emitted to air (Neto etal 2012)	0.25E-01	0
Impacts	**average impact**	**standard deviation on impacts**
Impacts FwEcotox PestLCI 2.0 (PAF·m3·day)	4.84E+02	1.94E+03
impacts FwEcotox Ecoinvent (PAF·m3·day)	1.52E+04	6.46E+04
impacts FwEcotox Neto et al. (PAF·m3·day)	1.26E+04	5.35E+04

In the present cases, the average of total emission fraction modelled with PestLCI 2.0 is more than 31 times lower than the total soil emission fraction estimated by Ecoinvent approach. The average PestLCI 2.0 modelled emission fraction to air is 3.4 times lower than the total emission fraction to air estimated by Neto et al. (2012) approach. This leads to differences in FwEtox estimates: 31 times lower with PestLCI model than Ecoinvent approach, and 26 times lower with PestLCI model than with Neto et al. (2012) approach. Very high standard deviations must be noticed on FwEtox, due to high differences in the ai.s' FwEtox potential.

4. Discussion

While having been intended mainly for arable crops, PestLCI 2.0 inventory model, due to its rather flexible framework, has here been adapted for viticulture without compromising the model framework. The calculation of toxicological impact potentials, beyond emission load quantification, was further complicated by ai.s not covered by the USETox^{TM} database, thus necessitating the calculation of CFs using the USETox^{TM} mode.

The case study results show that emission fractions vary in a large extent due to the ai.s properties and parcel and application conditions. High emissions fractions for most of the ai.s are compensated by very low application doses (cymoxanil, teraconazole) leading to moderate emissions (dose) but the ecotoxicological profile of the ai.s then has high importance on the final FwEtox results as shown by the FwEtox of Folpet and Aclonifen. Multiple factors differentiate the two case vineyards TMR1 and 3. The main factors are considered to be soil, sprayer equipment and type of pesticides applied. TMR1 shows higher emissions fractions than TMR3, however the total emission load is lower because of the low doses applied for some substances. Characterization through USETox^{TM} reveals different results for the two cases, mainly due to the high ecotoxicity potential of Folpet and to a lesser extent, of Aclonifen, even if both ai.s are applied via sheltered boom and tunnel sprayer (both limiting wind drift) (TMR1).

Inorganic pesticide ai.s, which are not modelled here, were also applied to the case vineyards: five applications for TMR1 including copper based ai.s and one application for TMR3 (Table2). The copper based pesticide ai.s are expected to further increase the FwEtox of TMR1 if included (Mackie et al. 2012; Vázquez-Rowe et al. 2012). A comparison of the present TMRs FwEtox profiles with the results obtained by (Vázquez-Rowe et al. 2012) with PestLCI 1 in Galician vineyards shows very good performance of the present TMR. Hence, TMR1's FwEtox is half of the lowest FwEtox mentioned by this author (Copper impacts removed) while TMR3's FwEtox is 400 times lower as this author's lower result.

Huge differences were found between the two static approaches (Ecoinvent and Neto et al. 2012) and PestLCI 2.0 based emission quantification. Accounting for the sole emissions that cross the parcel borders is a first element lowering the quantity of emitted pesticides as modelled by PestLCI 2.0, compared to the other approaches tested. However that is not the only cause of lower emissions and FwEtox; considering, for these emissions calculations, processes of evaporation, runoff and leaching, canopy influence, including the actual properties of the pesticide ai.s applied, of soils and sprayers allows for a more accurate adjustment of estimates to the real phenomenons. Degradation of pesticide ai.s and their uptake by the plants are actual processes that aren't considered in the static approaches tested, but accounted for in PestLCI 2.0 (Dijkman et al. 2012). The uptake of pesticides in crops including edible parts, is in PestLCI only considered as a fate/removal process and hence export from the field of pesticides via crops and crop residues is not considered nor quantified. Plant pesticide uptake processes are complex systems of coupled processes demanding dedicated models such as DynamiCROP (see e.g. (Fantke et al. 2011)).

In organic viticulture, sulfur and copper are the only means available to manage respectively powdery and downy mildew, and represent important quantities of applied pesticides. Sulfur and copper are not available in PestLCI 2.0 as the model is designed for modelling of organic pesticide ai. emissions. Thus, a comparison between conventional and organic viticulture or inclusion of organic managed cases in a study cannot be dealt with solely through PestLCI 2.0; a model similar to PestLCI is needed for inorganic pesticides. Considering the complex chemistry of inorganic chemicals, such a model will however have to deal with multiple species nature of inorganic chemicals in order to appropriately reflect the behaviour of these chemicals in soils.

PestLCI 2.0 could be improved by further developments of the accounting of airborne drift, which can be considerable (Jensen and Olesen 2014) but the complexity of the phenomena (Gil et al. 2008) and the lack of (generic) data are considered major obstacles for this improvement. More or less for the same reasons, pesticide metabolites are not accounted for in the present version of PestLCI 2.0. Accounting for application parameters as sprayers' speed, droplets size, temperature, relative humidity would be ideal for further refinement of the modelling of the spray mixture behaviour and fate, but these parameters are too difficult to obtain from the growers, and would further entail an even more complicated inventory.

High percentages of stones can be found in many vineyard soils; and modify water flow in the soil. These aspects could not be included in the present customization of PestLCI 2.0. However we recommend improvement of the way soil texture affects macropore transport in PestLCI 2.0 as an important issue to be considered in the coming PestLCI versions.

PestLCI 2.0, calculations must be carried out for each pesticide ai.s and application and thus remain a tedious task. Ideally, a possible connection with an external Excel database could potentially facilitate the automation (batch) of calculation and further pave the road for more detailed sensitivity analyses.

5. Conclusion

The PestLCI 2.0 customized version for viticulture, presented in the paper at hand, facilitates the calculations of emission loads for vertically trained vineyards with a wide range of sprayers and further provides a considerable PestLCI pesticide database update (even if non-exhaustive), of viticulture specific pesticides ai.s, and is complemented by the corresponding UseTOX^{TM} FwEtox characterization factors. A range of rather important specificities of viticulture that were not included in PestLCI 2.0. has been added like: i) specific sprayers drift characteristics, ii) the presence of a secondary canopy (cover-crop), iii) specific data like specific pesticide ai.s in the PestLCI database.

Application on two different case studies shows that emissions differ from a pesticide ai.s application to the other due to different emissions fractions related to environmental conditions and ai.s properties.

Huge differences were found between the two static approaches (Ecoinvent and Neto et al. 2012) and PestLCI 2.0 based emission quantification and FwEtox, The static approaches over-evaluate emissions and hence FwEtox from 2 to 31 times compared to PestLCI 2.0, due to different technosphere/ecosphere boundaries and accounting of degradation and plant uptake phenomena by PestLCI 2.0.

Some of the new PestLCI model parameters can also be used for other perennial or bush crops as long as equipment, shape of the canopy and pesticides a.i.s stay in the range of available options.

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Representing soil function in agriculture LCA in the Australian context

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ABSTRACT

Soils function is an important environmental value in Australia particularly for agriculture. However it has not been considered in life cycle assessment (LCA) studies to date in Australia, which has precluded its consideration alongside other environmental impacts. The agriculture sector and LCA community in Australia desire that soil function be captured in national life cycle inventory (LCI) and impact assessment frameworks. CSIRO has commenced a project to this end, with the first step being a workshop that brought together prominent soil scientists and LCA researchers to lay some groundwork for the project by i) prioritising the soil function and quality parameters of most importance and relevance for Australia, ii) considering how soil-related indicators can assign with evolving land use impact assessment frameworks, and iii) exploring the utilisation of spatial datasets to generate LCI for flows related to soil function to allow for more regionally-specific assessment of soil-related impacts.

Keywords: soil, LCI, LCIA, Australia, national inventory databases, agriculture, soil carbon, erosion, acidification

1. Introduction

This paper describes the outcomes from an Australian workshop, at which a proposed approach for integrating indicators of soil function and quality into the Australian national life cycle inventory database was developed.

Soil function and qualities and their influence on productivity and ecosystem services have been under-represented in LCA to date. Recent efforts by the UNEP SETAC Life Cycle Initiative have devised an impact assessment framework that captures the eco-system service functions of land, including soil functions (Koellner et al. 2013). However this has yet to translate into the representation of soil-related impacts in life cycle studies.

Soil-related impacts are pertinent to many production systems that involve transformation of land. However interest in soil impacts has been expressed most keenly by those working on agriculture-based production systems. This is because soil function is inherently linked to agricultural productivity, and the long-term sustainability of agriculture relies on protecting and enhancing it. Hence there is more incentive to influence the protection of soil in agriculture than in any other sector.

The absence of soil function considerations in LCA to date is in contrast to the recognition of soil-related problems by the agriculture sector. Compared to soils on other continents, Australia’s soils are very old, highly weathered and relatively infertile (DAFF, 2014). While there are areas of highly fertile soil, Australia’s soils often have poor structure with low levels of organic matter and are affected by salinity, sodicity, and acidification, and in some regions are subject to wind and water erosion. So, for Australia in particular, declining soil quality is a major concern. The level of investment by the agriculture sector and governments to rectify soil-related impacts in Australia (estimated at A$124 million in 2011; DAFF 2014) is higher than for other environmental impacts such as eutrophication and pesticide impacts on sensitive environments and global warming.

Therefore the agriculture sector and LCA community in Australia desire that soil function and qualities be captured in national inventory and impact assessment frameworks. The Commonwealth Scientific and Industrial Research Organization (CSIRO) commenced a project to this end, with the first step being a workshop that brought together soil scientists and LCA researchers to lay some groundwork for the project.

This project follows on from a prior program of work completed in 2013 to establish a national life cycle inventory (LCI) dataset for Australian agricultural processes (AusAgLCI) (Eady et al. 2013) which is available within the AusLCI database (alcas.asn.au/AusLCI). The developed data sets currently contain inventory that allows for the assessment of resource depletion, global warming, eutrophication, acidification, and human- and eco-toxicity. The current project aims to extend the inventory to enable the assessment of soil-related impacts.

2. Methods

A two-day workshop was held in Canberra on the 8 and 9 April, 2014, which was attended by around 50 soil scientists, LCA researchers and, government and research investment managers (25 in person and 25 by webinar) (see www.alcas.asn.au/events/roundtables). The aim of the workshop was to connect LCA researchers with Australian soil scientists, for the purpose of reviewing the latest science related to soil processes in Australia and considering how best to characterise soil-related impacts in LCA in the Australian context.

After some introductory sessions on LCA and international developments in land use impact assessment, the forum heard presentations from prominent Australian soil scientists about notable soil characterization developments in Australia. The forum also connected via webinar with overseas LCA researchers (from Europe, North America and New Zealand) to gain insights into international developments and interest in this field.

The soil issues covered in the workshop were those for which there is active and significant research taking place in Australia, and which have been identified as the key soil condition indicators (Australian Government, 2011). They were compaction, soil organic carbon, erosion, contamination, soil biota and acidification. While not covered in the structured presentations, the additional issues of soil salinization and structural decline were also discussed.

Based on the structured discussion at the workshop it was possible to i) nominate the soil function and quality parameters of most importance and relevance for Australia, ii) propose how soil-related indicators of interest in Australia can align with evolving impact assessment frameworks, and iii) identify how Australian spatial datasets can be utilised to generate LCI for soil-related flows to allow for more regionally-specific assessment of soil-related impacts.

3. Results and discussion

3.1. Priority soil function and quality parameters for Australia

The geochemical processes that influence soil characteristics vary in different parts of the world, with a general distinction drawn between the high latitude countries where glacial weathering has a significant renewal effect on soil quality or those areas where significant volcanism or annual flooding has rejuvenated landscapes, and continents such as Australia where under the influence of flat topography and long periods of weathering, low rainfall and wind erosion, most soils have lower resilience and are more vulnerable to degradation. This means there may be differences in the way soil-related impacts are best represented in Australia (and other similar geological regions e.g. parts of Africa) compared with Europe, which has been the context for life cycle impact assessment development to date. Therefore, it was worth considering the soil function and quality parameters that are of relevance and importance for Australia, to ascertain if they can be adequately captured in evolving land use impact assessment frameworks. Furthermore, the practicality of collecting data for these parameters for inventory development needs to be considered.

Participating soil scientists were asked to rank the soil characteristics according to their environmental importance in Australia and the expected difficulty of inventory development. The results of participant feedback are shown in Figure 1.

The priority soil attribute raised at the workshop (those in the top left-hand quadrant of Figure 1) were consistent with those identified in Australia's State of the Environment reporting (Australian Government 2011) and the national soil R&D strategy (DAFF 2014). The leading challenges are acidification, erosion and loss of soil carbon. This degradation will increasingly affect Australia's agriculture unless carefully managed, and in some cases ameliorated. Assessment of soil condition for regions with cropping and/or intensively managed grazing systems (approximately 75 million hectares) showed that for:

– **acidification**—50 per cent of regions were in very poor or poor condition, while 95 per cent of regions showed a deterioration in soil pH
– wind and water **erosion**—53 per cent of regions were assessed as in poor condition, while soil cover in 11 per cent of regions was deteriorating
– **soil carbon**—33 per cent of regions had very poor or poor soil carbon content, while 85 per cent of regions had ongoing deterioration in soil carbon content.

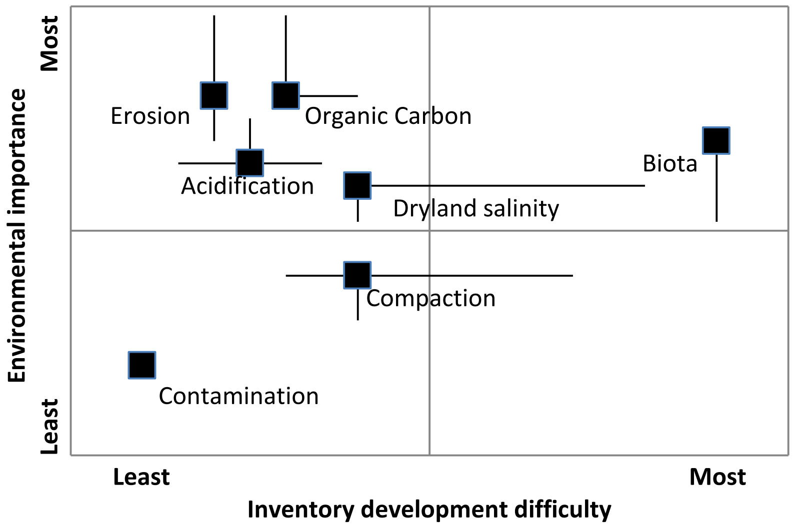

Figure 1. Ranking of environmental importance and difficulty of inventory development (with the horizontal and vertical bars indicating spread in opinion).

Dryland salinity has also been identified as an ongoing threat, although the millennium drought may have slowed its spread temporarily. However, salinity is less easily linked to on-site agricultural management. While its expression (in contaminated soil or near-surface groundwater) can adversely impact on agricultural production, its cause may be substantially removed in space and time (e.g. clearing or irrigation elsewhere in the catchment at other times) and the risk components are complex (Grundy et al. 2007). Compaction was assessed to be of lesser overall importance while diverse and high populations of soil biota were considered to be important but difficult to quantify. Soil contamination was also thought to be less of an issue as it is mostly related to point source pollutant emissions and hence quite localized, and impacts can be adequately captured in existing inventory and impact assessment (e.g. heavy metals in fertilizers).

In relation to the practicality of inventory development, it was considered that datasets of sufficient representativeness exist for soil organic carbon, erosion and acidification. For organic carbon, data on stocks and flows are available from cropping systems models such as APSIM, continent scale models such as FullCAM, and geospatial modeling (Viscarra Rossel et al. 2014). For erosion, there are model data assimilation approaches such as those used by Chappell et al. (2013) to estimate net rates of erosion. For acidification, data are available from the Australian Soil Resource Information Systems (www.asris.csiro.au). For salinity and compaction more work may be required to adapt existing datasets; detailed data are available for areas of salinity high risk e.g. the south-west cropping lands (Furby et al. 2009). There is a considerable paucity of measurement methods available for soil biota at the spatial scale required for national inventory making it impractical at this time to consider including biota-related inventory.

It was noted that in the Australian context the sustainability objectives for soil function and quality were both the maintenance of eco-systems services and the maintenance of the soil resource itself. As will be discussed further in the next section, existing LCA frameworks are mostly concerned with the eco-system services function of soils. Soil is not captured by resource depletion indicators in LCA as fossil fuels and minerals are. In the Australian context avoiding the depletion of soil resources is important. This is because the natural reference state for land in Australia is already highly eroded with very shallow soil profiles and high propensity for ongoing erosion. Consequently it is easy for tipping points to be reached.

In summary, the set of characteristics considered to be important because they significantly influence the productivity, the soil resource base, and ecosystem services, they are related to land management in the agricultural system and for which there are reasonable data available, are soil organic carbon, erosion and acidification. These will become the priorities for further investigation in the CSIRO project.

3.2 Alignment with land use impact assessment frameworks

The alignment of inventory with impact assessment is an important consideration in developing regional inventory datasets. The authors appreciate the challenges of regionalized impact assessment and the need to harmonize impact indicators (Saad et al. 2013), and acknowledge that Australia-specific inventory needs to enable impact assessment using internationally consistent indicators and impact assessment methods.

A good starting point is the guideline for land use impact assessment developed under the UNEP/SETAC Life Cycle Initiative (Brandão and Canals 2013; Koellner et al. 2013; Saad et al. 2013). It is the most developed model relating to land use to date, and proposes a characterization model that links land characteristics with indicators of eco-system services. Figure 2 presents an extract from this model showing the cause and effect chains related to soil characteristics and including the midpoint indicators proposed to date.

Overlaid on this model are the priority soil characteristics identified for the Australian context (shaded in light grey in Figure 2). Two of these, soil organic carbon and erosion, are already captured in the land use impact assessment model (light grey). Soil organic carbon (tC/ha) is proposed in the model as a proxy for fertility, influencing biotic production (Brandão and Canals 2013). It is also the indicator used to quantify soil carbon sequestration. Erosion regulation potential is captured in the model, with the proposed proxy indictor being erosion resistance potential (t soil/ha/yr) (Saad et al. 2013).

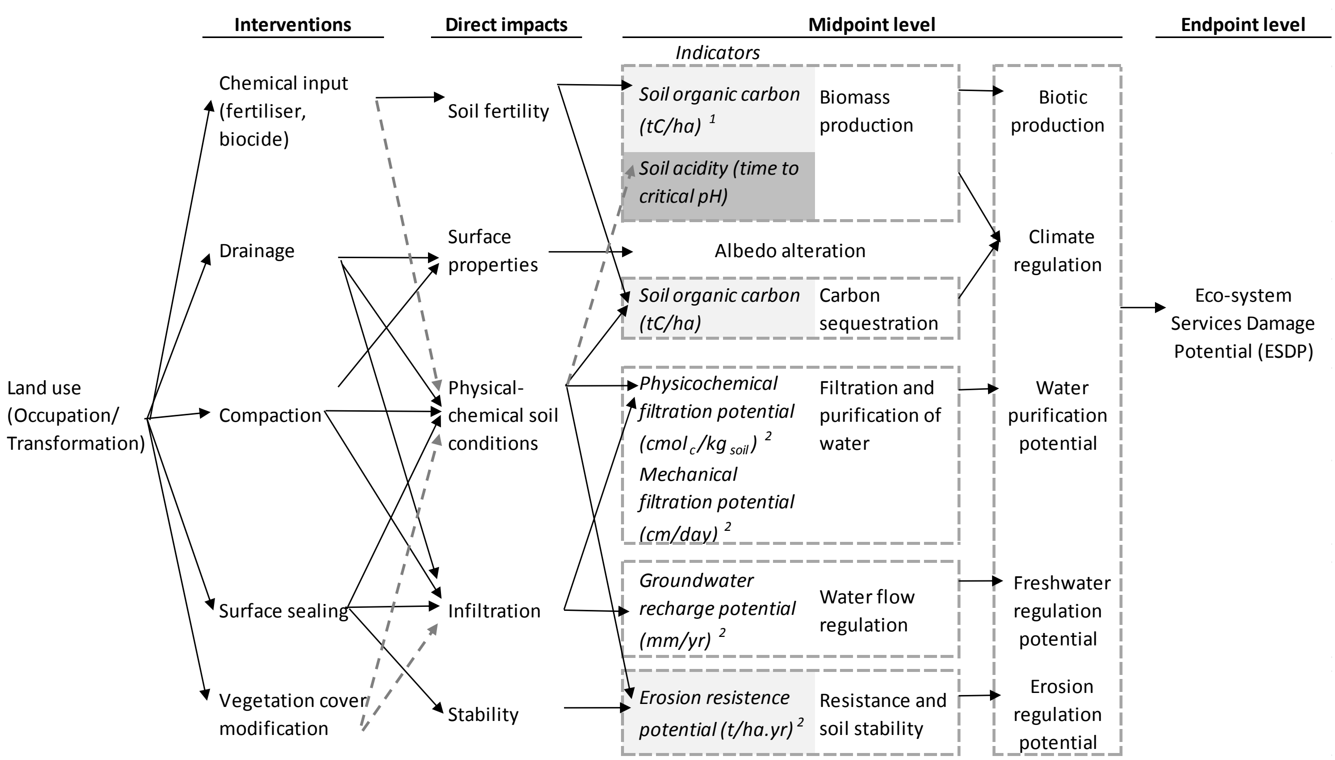

Figure 2. Extract from the land use impact assessment model developed under the UNEP/SETAC Life Cycle Initiative (Koellner et al. 2013), and incorporating the midpoint indicators proposed for biomass production (1) (Brandão and Canals 2013), and other eco-system services (2) (Saad et al. 2013). The shaded indicators are those identified as important in the Australian context.

Soil acidity is not captured in the model (dark grey in Figure 2). It adversely influences agricultural productivity in Australia (Australian Government 2011) and could be considered as a moderator of biotic production along with soil organic carbon. Some consideration may need to be given to how it can be integrated into the model.

In summary, soil-related inventory developed for Australia for soil organic carbon and erosion could align with the emerging land use impact assessment framework. Even though compaction is not currently a priority for inventory development, it could also be accommodated at a later date as midpoint indicators reflecting the impacts of compaction have been proposed (physicochemical and mechanical filtration potentials and groundwater recharge potentials) (Saad et al. 2013). An adaptation to the impact assessment framework to accommodate Australian conditions, could be the inclusion soil acidity as a moderator of biotic production. Furthermore, including soil within abiotic resource depletion impact categories would allow for consideration of Australia's soil resource conservation objective.

3.3 Utilisation of spatial datasets to generate LCI for flows related to soil function

The spatial scale of the agricultural LCI datasets (AusAgLCI) is at the level of agro-ecological regions (Williams et al. 2002). To enable cost effective inventory development across the breadth of Australia's agricultural regions, GIS-enabled spatial data are utilized. As previously noted, spatially-linked data for soil organic carbon, erosion and acidification are available, and it is proposed that these be used to supply soil-related inventory (see an example in Figure 3).

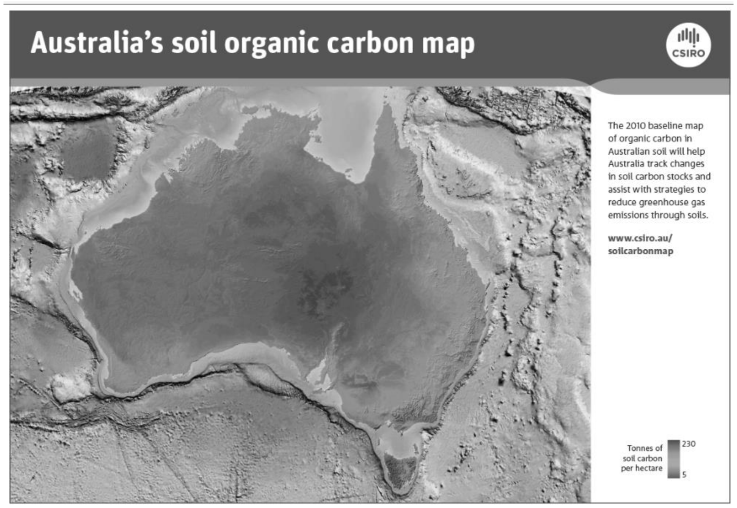

Figure 3. The 2010 baseline map of soil organic carbon in Australian soil; a base to track changes in soil carbon stocks and assist with strategies to reduce greenhouse gas emissions through soils (Viscarra Rossel et al. 2014).

As part of the UNEP/SETAC land use impact assessment framework generic characterization factors for global application have been developed, which characterize the mid-point indicator per unit of land, based on regionally-disaggregated modeled estimates (Brandão and Canals 2013; Saad et al. 2013). For the purposes of extending Australian agricultural inventory, region- and production system-specific inventory flows derived from the spatial data are proposed to be employed rather than defaulting to the generic characterization factors. These will give a more robust and representative estimate of the key soil-related impacts in the Australian context. Spatial data for soil organic carbon, acidity (pH) and net soil erosion already exist, and are represented as 'stocks'. The challenge will be to couple these estimates of 'stocks' with a means of estimating 'flows' (stock changes), and to link flows to management practices and interventions.

A similar approach was taken in the previous inventory development project (Eady et al. 2013) to generate region- and production system-specific pesticide flows by using PestLCI to partition flows between soil, air and water compartments. This meant that more robust regional inventory is possible, rather than relying on generic characterization factors embodied within the impact assessment methods, that do not account for the climate and soils of the region. Hence, we propose to take a similar approach for soil attributes and use the best existing data to describe mid-points indicators.

4. Conclusion

An approach for extending Australian agricultural LCI datasets to include soil-related flows has been proposed, which can be consistent with evolving international frameworks for land use impact assessment. Spatially-linked data for soil organic carbon and erosion are available to generate inventory flows that align with proposed midpoint indicators for biotic production and erosion regulation. Acidification is a major constraint to soil function in the Australian context, and a way of adapting the biotic production mid-point indicator to be moderated for acidification may need to be considered. The tasks for the project will be to convert the ‘stock’ information available from the datasets into ‘flows’ (stock changes) for the inventory, and to link the flows to management practices and interventions. The inclusion of soil resources within adiabatic resource depletion impact categories would also allow for consideration of Australia’s particular soil resource conservation objectives.

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Estimating Energy and Greenhouse Gas Emission Savings through Food Waste Source Reduction

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ABSTRACT

To help advance the source reduction of food waste in the U.S. municipal solid waste (MSW) stream, ICF International is working with the U.S. Environmental Protection Agency to develop life-cycle energy and greenhouse gas (GHG) emission factors for food waste to compare the impacts of source reduction and disposal options for common food waste types in the U.S. MSW stream. ICF has developed cradle-to-retail source reduction energy and GHG emissions factors as part of EPA's Waste Reduction Model (WARM) for four food waste types: grains, bread, fruits and vegetables, and dairy products. These factors are expected to have broad appeal across the food service, restaurant, and materials management industries and governmental organizations and are expected to be used as part of EPA's Food Recovery Challenge to encourage reduction and recovery of food waste.

Keywords: food waste, greenhouse gases, source reduction, organics

1. Introduction

In 2012, food waste was the largest single component in the U.S. municipal solid waste (MSW) stream, comprising over 21% of MSW discards my mass (U.S. EPA 2014a). The U.S. Environmental Protection Agency's (EPA) food recovery hierarchy identifies source reduction as the most preferred method for addressing food waste (U.S. EPA 2014b). In the context of MSW, source reduction is defined as the reduction of waste generation at its source by means of altering the waste generation source (e.g., via more efficient consumer use of materials, reuse, or improved product design). In the case of food waste, source reduction refers to upstream reductions in food production. It can result from any activity (e.g., lowering consumption, reducing food spoilage, avoiding disposal of uneaten food) that reduces the amount of an agricultural input needed and therefore used to make food. As a result, there is demand for a simple tool allowing waste managers, food service managers, and other stakeholders to estimate the energy and greenhouse gas (GHG) emissions avoided through food waste source reduction. To help meet this demand, ICF International worked with EPA to develop life-cycle energy and GHG emission factors for food waste to compare the impacts of source reduction and other disposal options for common food waste types in the U.S. MSW stream using EPA's Waste Reduction Model (WARM).

EPA created WARM to help solid waste planners and organizations track and voluntarily report GHG emissions reductions from several different materials management practices. WARM calculates and totals the relative GHG emission and energy impacts of baseline and alternative materials management practices—source reduction, recycling, combustion, composting, and landfilling—using GHG emission factors that EPA has developed based on a materials life-cycle approach. ICF has provided EPA with ongoing support for the development of life-cycle emission factors and in the development of and updates to WARM since its creation.

WARM currently models life-cycle GHG impacts for over 50 material types. Prior to the development of the energy and GHG factors described in this paper, WARM included a general "food scraps" waste category representative of national average food waste composition. For the "food scraps" category, WARM modeled energy and GHG impacts from three end-of-life pathways—landfilling, combustion, and composting—but did not consider GHG emissions associated with raw materials acquisition, transportation, and processing of food products before they enter the U.S. MSW stream. Therefore, it was not possible to use WARM to estimate the energy and emissions avoided through reduced production of food, also known as source reduction.

To better understand and reduce the impacts from this growing waste stream, ICF worked with the EPA to expand WARM by developing life-cycle energy and GHG emission factors for food waste to include cradle-to-retail upstream energy and emission factors for four of the largest food waste types in the U.S. MSW stream (bread, grain products, dairy products, and fruits and vegetables) and a weighted average for mixed food waste. This paper summarizes the methods and results for developing the factors for these food waste types suitable for inclusion within the existing scope and boundaries of WARM.

2. Methods

When a food product is source reduced, GHG emissions associated with producing and transporting the product and managing the production and post-consumer waste are avoided. Consequently, source reduction of food waste provides GHG emission benefits by: (1) avoiding the "upstream" GHGs emitted in the raw material acquisition, manufacture, and transport of the source-reduced material; and (2) avoiding the downstream GHG emissions from waste management. This paper describes the development of energy and GHG emission factors to estimate the avoided upstream GHG emitted in the raw material acquisition, manufacture, and transport of source-reduced food waste using a methodology consistent with existing WARM factors.[1]

2.1. Scope and Boundaries

Food waste types to be modeled in WARM were selected based on: (1) their usefulness and practicality to WARM users; (2) the share of total U.S. food waste the materials would individually and collectively comprise; (3) the availability of relevant, high-quality LCI data; and (4) the practicality of emission factor development. U.S. Department of Agriculture (USDA) Economic Research Service (ERS) loss-adjusted food availability data from 2010 were used to determine the food types constituting the largest share of the U.S. MSW stream (USDA 2012b). As summarized in Table 1, ICF evaluated the share of food waste generated in the United States for five food groups: grain products, fruits and vegetables, red meat, poultry, and dairy products. For each of these groups, ICF chose to model one or more food types that would cumulatively represent at least half of the waste generated for that group. This paper presents the methods for developing source reduction factors only for the basic categories of grains, fresh fruits and vegetables, and dairy. Future work will address source reduction factors for beef and chicken.

Table 1. Share of Total 2010 U.S. Food Waste Stream Represented by Materials Modeled in WARM.

Food types modeled in WARM		**Share of U.S. food waste in 2010**	**Share of weighted average food type in WARM**
Grains	Wheat	5.3%	68.3%
	Corn	1.3%	16.8%
	Rice	1.2%	14.9%
	Total	*7.8%*	*100%*
Fruits and vegetables	Potatoes	8.1%	27.5%
	Tomatoes	7.9%	27.0%
	Citrus	6.1%	21.0%
	Melons	2.7%	9.3%
	Apples	2.4%	8.2%
	Bananas	2.0%	6.9%
	Total	*29.3%*	*100%*
Red meat	Beef	5.5%	100%
Poultry	Chicken	6.5%	100%
Dairy	Fluid milk	7.3%	60.3%
	Cheese	1.2%	12.0%
	Yogurt	0.6%	5.4%
	Other dairy	1.6%	22.3%
	Total	*10.9%*	*100%*
All food materials modeled in WARM		61.4%	

Source: USDA 2012b.

The scope of the food waste source reduction energy and emission factors included in WARM is cradle-to-retail, meaning that the factors include the upstream impacts of all unit processes prior to retail storage and consumer use. The main GHG emission sources considered in the source reduction factors include: (1) the upstream production of agricultural inputs, such as fertilizer, irrigation, livestock feed, and fuel; (2) on-farm energy use; (3) on-farm non-energy emissions, including fertilizer application, manure management, and enteric fermentation; (4) some energy from food processing, depending on food type; (5) energy use and fugitive emissions from refrigeration during processing and transport of some materials; and (6) retail transportation energy. In order to

[1] Full documentation for existing WARM factors is available at http://www.epa.gov/warm.

maintain consistency with other material categories in WARM, the impacts of consumer transport and material use are not included. Furthermore, these downstream emissions are outside the scope of "source reduction" as defined by the U.S. EPA.

Several key exclusions from the scope include: (1) infrastructure and capital equipment; (2) induced land-use change; (3) production, use, and end-of-life management of packaging materials; (4) energy use, fugitive refrigerants, and food loss rates at retail locations; and (5) consumer transport, refrigeration, and energy use for cooking. In most cases, these exclusions were made to ensure consistency with existing WARM energy and GHG emission factors or because impacts were considered negligible. WARM includes energy and GHG emission factors for the production and disposal of most common food packaging materials, including corrugated cardboard, other paper types, plastic resins, and glass. Packaging material wastes and food wastes are commonly managed using different waste management pathways (e.g., recycling of paper, plastics, and glass). Therefore, this aspect of the food life cycle was excluded from the food waste energy and GHG emission factors in order to avoid redundancy between the food waste factors and existing packaging material factors available within WARM and to more-accurately capture the end-of-life impacts from management of packaging materials.

Due to data limitations, certain differences in boundaries exist between the factors for different food waste types in addressing food processing. The dairy material type includes energy and GHG emissions associated with all processing steps for the main dairy-based products consumed in the United States, include additional upstream processing to freeze or otherwise modify the flavor or texture prior to sale. The boundaries of the grains material type includes energy used for grain drying prior to product-specific processing, but exclude processing steps such as milling, packaging and cooking due to unavailability of national average grains processing data for wheat, rice, and corn. However, ICF has developed a separate source reduction factor for wheat-based bread to account for the energy and emissions associated with milling and baking. Similarly, because all of the components included in the fruits and vegetable factors can be consumed as fresh fruits and vegetables and due to the lack of data on fruit and vegetable processing, ICF has assumed that all fruits and vegetables enter the waste stream as fresh fruits and vegetables.

The material-specific food waste source reduction factors in WARM are linked to existing end-of-life factors for landfilling, combustion, and composting of general food waste to cover the full cradle-to-grave life cycle of food products. While future updates to WARM will include the development of material-specific food waste disposal factors for landfilling, combustion, and composting, the development of such factors was outside the scope of the effort described in this paper. Future updates to WARM will also include modeling the energy and GHG emissions from anaerobic digestion of food waste.

2.2. Functional Unit

The functional unit for all food waste material types is one tonne of food at end-of-life.

2.3. Data Sources and Representativeness

ICF gathered LCI data for primary and secondary inputs to production of grains, bread, fruits and vegetables, and dairy products from industry, academic, and government sources, including LCI databases. ICF assessed the scope and boundaries of each data source to identify any smaller data gaps requiring secondary data and to determine appropriate boundaries for the upstream food waste factors that are both consistent across all food waste types and consistent with existing WARM methodology. A summary of the key primary LCI data sources is provided in Table 2.

Table 2. Key Primary LCI Data Sources for Food Types Modeled in WARM

Food types modeled in WARM		Data Source
Grains	Wheat Corn Rice	USDA Undated.
Bread	Wheat	USDA Undated; Espinoza-Orias 2011
Fruits and vegetables	Potatoes	Ecoinvent Centre 2007.
	Tomatoes Citrus Melons Apples	UC Davis 2013.
	Bananas	Luske 2010.
Dairy	Fluid milk Cheese Yogurt Other dairy	Thoma et al. 2010.

In developing food waste energy and GHG emission factors for WARM, ICF sought to use data sources that were most representative of current, national average practices with a goal of representing the main components of the U.S. MSW stream. All primary data sources are based on data collected in 2001 or later, with most based on data from 2008-2012. In order to develop national average source reduction energy and emission factors, several key assumptions were made. First, ICF assumed that all of food types modeled would be produced in the United States, with the exception of bananas[2], and would be produced using conventional (i.e., non-organic) farming practices. Secondly, the differences in production impacts across different breeds, varieties, or types components of each food waste category were not considered in the analysis. For example, LCI data for the production of Fuji apples were assumed to be representative of all apple production in the United States. Likewise, LCI data for the farming of oranges was assumed to be representative of all citrus production due to lack of data for production of other citrus fruits and food consumption data showing that oranges comprise 65% of citrus fruits consumed in the United States in 2012 (Boriss 2013).

2.4. General Methodology

A similar methodology was applied for developing the energy and GHG emission factors for all food waste categories. Any differences in methodology are identified in each of the material-specific subsections below. In general, the emissions were calculated in two separate stages: first, energy-derived emissions were calculated by determining the cumulative energy demand for producing one kilogram of each food type. Secondly, non-energy emissions were estimated and added to the fossil fuel-derived emissions.

To estimate the energy-derived emissions, ICF calculated the cumulative energy demand for each dataset within SimaPro through the Ecoinvent version 2 cumulative energy demand impact assessment method provided in the software. This method calculated the total life-cycle energy in MJ required to produce one unit of food product and then separated the total into several categories, including: petroleum, nuclear power, biomass, natural gas, coal, and renewables. Each energy source's contribution to the total energy demand was then multiplied by the fuel-specific carbon coefficients used in WARM for all materials to determine the total energy-derived emissions associated with the production of one unit of food product. The caloric biomass energy embedded in the food products themselves were excluded from the analysis because they did not contribute to the energy-related emissions associated with their manufacture.

Non-energy emissions were calculated using a variety of methods, depending on the emissions source, the food type, and the LCI data available. A summary of non-energy emissions methodology is provided in each material-specific subsection below. For all food types, ICF utilized the IPCC Tier 1 method for managed soils to calculate the total amount of N_2O and CO_2 released from fertilizer application, run-off, volatilization, and leaching (IPCC 2006).

[2] Foreign-grown bananas were included within this assessment because they are one of the largest sources of fruit and vegetable waste within the U.S. waste stream. They were assumed to be produced in Central America using conventional farming practices due to the lack of suitable climate for their cultivation on a large scale within the United States.

Additional detail on the methodology for developing food waste source reduction energy and GHG emissions factors is provided as part of the WARM Version 13 documentation chapter for Food Waste, available at http://epa.gov/epawaste/conserve/tools/warm/SWMGHGreport.html.

2.5. Grains and Bread Methodology

The grains energy and GHG emission factors includes milling of wheat into flour but assumes that wheat flour, corn, and rice can be purchased as dried grains without further processing or cooking. The emission factor for grains may understate the upstream emissions associated with corn and rice products that have undergone further processing. Therefore, ICF developed the bread factors to supplement the grain factors by including the additional energy used to process wheat flour into bread, which is the predominant use for wheat flour (USDA 2012a).

ICF identified recent LCI data for the three grain products—wheat, corn, and rice—available in the USDA National Agricultural Library's LCA Digital Commons database. The Digital Commons database is intended to provide LCI data for use in life-cycle assessment (LCA) of food, biofuels, and a variety of other biological products. All datasets in the LCA Digital Commons database are specific to grain production in U.S. states. The LCI data from the Digital Commons datasets only provide material inputs, outputs and, processes in units of magnitude per unit of agricultural product produced without any estimates of the energy or GHG impacts associated with production. For example, the LCI data include estimates of the amount of fertilizers needed for grain production but do not include data on the energy needed to for fertilizer production or the direct GHG emissions from fertilizer application. In order to translate these values into the actual energy demand and emissions associated with agricultural production, ICF identified matching unit processes and corresponding LCI data for those materials and processes within the life-cycle software, SimaPro. The unit processes within the database are taken from the Swiss Ecoinvent version 2 database and the U.S. LCI Database. Several life-cycle unit processes from Ecoinvent use European electricity grid assumptions. Therefore, ICF updated the LCI data for these unit processes within SimaPro to utilize U.S. electricity grid assumptions using the updated US-EI database update for SimaPro prepared by EarthShift.

ICF identified the most representative and recent state-level datasets for winter wheat, corn, and rice within the Digital Commons database. Representativeness of state-level data in the Digital Commons database was determined through a National Agriculture Statistics Survey (NASS) commodity data search of the production levels of each grain on a state by state basis. Arkansas and Kansas datasets were chosen for rice and winter wheat production, respectively, because they were the largest producers of those grains. For corn, due to the similarly large shares of corn produced by both Iowa and Illinois, ICF created a weighted average from datasets from both states.

The Digital Commons LCI data assumes that the production of each of the three grains included in WARM leads to the production of one or more co-products. These co-products include corn silage, corn stover, wheat straw, and rice straw. The amount of each of these co-products produced per ton of grain ranges from 0.04 tonnes for corn silage to 0.58 tonnes for wheat straw. Rather than be treated as a waste, each co-product has an economic value and therefore should be allocated some percentage of the total energy and emissions required for grain production. In keeping with ISO 14044 standards, ICF first investigated avoiding allocation by expanding the system analyzed to include the impacts from co-product use and disposal (ISO 2006). However, due to the prohibitive time and data requirements for system expansion, ICF chose to allocate impacts to co-products in proportion to the economic value of the products. Using data from the USDA ERS Commodity Costs and Returns database, ICF determined the economic value per acre of production for corn, corn silage, rice, wheat, and wheat straw for each of the LCI data years (USDA 2013a). This provided sufficient data to determine economic allocation percentages for wheat and wheat straw. Supplementary data from a 2009 study by van der Voet et al. provided prices for corn stover, allowing us to estimate the allocation percentages for corn, corn silage, and corn stover. However, ICF was unable to find a reliable source for the economic value of rice straw. An anecdotal article cited rice straw's value at approximately $10 to $20 per acre, which would translate to allocation of 1 to 3% of rice production energy and emissions to rice straw (Smith 2004). Due to the lack of reliable data and the likely low economic value of rice straw, ICF allocated 100% of energy and emissions to rice grain. These allocation calculations are summarized in Table 3.

Table 3. Allocation Assumptions for Grains Co-Products

Grain	Primary Product/Co-Product(s)	Weighted Average Economic Value across LCI Data Years
Wheat	Primary product: wheat grain	96.1%
	Co-product: wheat straw	3.9%
	Total	100%
Corn	Primary product: corn grain	85.8%
	Co-product: corn silage	0.6%
	Co-product: corn stover	13.6%
	Total	100%
Rice	Primary product: rice grain	100%
	Co-product: rice straw	0%
	Total	100%

The final results for each grain were calculated by summing the energy emissions and non-energy emissions and then combining each grain into a weighted emission factor based on production in the selected states during their respective data years. From there, the three grain emission factors were combined into a final weighted emission factor based on the relative shares of each within the U.S. municipal solid waste stream. Retail transport energy and emissions were estimated with the Bureau of Transportation Statistics commodity flow survey, consistent with other materials in WARM, and are equal across the three types of grains.

Energy and GHG emissions from bread production were estimated by taking an estimate of bread production energy intensity from Espinoza-Orias et al. 2011, which contained LCI data characterizing the energy use associated with producing bread. For the purposes of this analysis, white bread was chosen as it is more common than wheat bread. The study found that wheat milling and baking, respectively, had energy demands of 0.059 kWh and 0.600 kWh per loaf of bread, which was assumed to be 0.8 kg. This equated to 2.66 MJ of cumulative energy demand to prepare one tonne of bread, of which the entirety was assumed to be taken from the national average electricity grid. To estimate the total farm-to-retail energy associated with bread, ICF summed the bread production energy emissions with those for wheat flour, but did not include corn or rice. Corn and rice were excluded from this process because the energy use data for milling and baking were based on wheat bread production and because wheat-based bread is the predominant bread category in the United States (USDA 2012a).

2.6. Fruits and Vegetables Methodology

LCI data used to develop the energy and emission factors for fresh fruits and vegetables in this memo came primarily from three sources. Data for the production of apples, melons, tomatoes, and oranges came from the University of California Cooperative Extension's (UCCE) sample cost production studies (Fake et al. 2009, O'Connell et al 2009, Stoddard et al. 2007, Wunderlich et al. 2007). These studies are intended as hypothetical guides for farmers to produce crops, and include yield projections and sample requirements for fuel, fertilizers, irrigation, and plant protection products. Data for the production of bananas was acquired from a 2010 LCA conducted by Soil and More International, on request of the Dole Food Company (Luske 2010). The banana LCA study characterizes the cradle-to-retail GHG emissions associated with banana production in Costa Rica and retail in Western Europe. In developing the source reduction energy and emission factors, ICF used supplementary data to model international shipping and retail transport to the United States. Lastly, the data for potato production was acquired from the Ecoinvent version 2 database, available within the SimaPro LCA Software.

In assessing energy and GHG emissions from production of bananas, ICF supplemented the primary data source with secondary data and applied a different methodology to maintain consistency with the other fruits and vegetables within the weighted emission factor and with the scope of WARM. First, to narrow the scope of the data to cradle-to-retail, ICF did not assess the impacts of retail storage at the destination country. Secondly, to make the dataset more relevant to bananas sold within the United States, ICF assumed an average transportation distance from Central American banana plantations to U.S. ports, acquired from a separate study on fruit transportation distances (Bernatz 2009).

Unlike the other components of the fruit and vegetable energy and emission factors, bananas are shipped internationally in specially-made, refrigerated cargo containers to prevent over-ripening prior to sale. The average transportation distance to the United States was multiplied by a separate factor for emissions per ton-kilometer of refrigerated ocean cargo transport (BSR 2012). Additionally, due to the role of refrigeration in the ocean transport of bananas, ICF incorporated Luske 2010's estimate of fugitive refrigerant emissions during processing and transport.

For this analysis, distribution of fruits and vegetables to their final point of sale was assumed to have two components: the energy and GHG emissions associated with fossil fuel combustion from vehicle operation in addition to the GHG impact of fugitive refrigerants emitted from refrigerated vehicles. The GHG emissions from vehicle operation were a product of diesel fuel combustion. Fugitive emissions of refrigerants consisted of a mix of 1,1,1,2-Tetrafluoroethane (R-134a), Chlorodifluoromethane (HCFC-22), Monochloropentafluoroethane (R-155), and 1,1-Difluoroethane (HFC-152a). Due to lack of data for fruit and vegetable-specific transportation, the fugitive emissions associated with refrigerated vehicle transport were assumed to be the same as for refrigerated dairy delivery via a medium-sized truck (Thoma et al. 2010). In the Thoma et al. 2010 study, estimates of fugitive emissions of refrigerants during the transport phase were estimated via a sales-based approach, which equated purchases of refrigerants for the truck fleet to fugitive refrigerants released via leakage.

For this analysis, distribution of fruits and vegetables to their final point of sale was assumed to have two components: the energy and GHG emissions associated with fossil fuel combustion from vehicle operation and the GHG impact of fugitive refrigerants emitted from refrigerated vehicles. The GHG emissions from vehicle operation were a product of diesel fuel combustion. Fugitive emissions of refrigerants consisted of a mix of 1,1,1,2-Tetrafluoroethane (R-134a), Chlorodifluoromethane (HCFC-22), Monochloropentafluoroethane (R-155), and 1,1-Difluoroethane (HFC-152a). Due to lack of data for fruit and vegetable-specific transportation, the fugitive emissions associated with refrigerated vehicle transport were assumed to be the same as for refrigerated dairy delivery via a medium-sized truck (Thoma et al. 2010). In the Thoma et al. 2010 study, estimates of fugitive emissions of refrigerants during the transport phase were estimated via a sales-based approach, which equated purchases of refrigerants for the truck fleet to fugitive refrigerants released via leakage. Retail transport ton-miles per shipment for all fruits and vegetables were informed by the Bureau of Transportation Statistics (BTS) 2007 Commodity Flow Survey (BTS 2010). Bananas were assumed to have land-based domestic transport in addition to refrigerated ocean transport described above.

Retail transport of perishables such as fruits and vegetables also results in losses due to spoilage and physical damage to the produce that would render it unfit for sale. Loss rates for the transport of fresh fruits and vegetables from production to retail were derived from USDA Economic Research Service (ERS) loss-adjusted food availability data (USDA 2012b). Loss rates for each fruit and vegetable in the analysis were compiled from USDA (2012b) and then re-weighted based on each product's share of the waste stream. The loss rates were specific to losses incurred strictly during the transport of fresh fruits and vegetables instead of a weighted mix of fresh and processed fruits and vegetables in order to maintain consistency with the scope and methodology of the analysis. The calculated weighted loss rate of 7.1 percent was applied to both production and transportation emissions of all fruits and vegetables in the study.

Impacts from co-products were not included in this analysis due to data limitations. For apples, oranges, melons, and tomatoes, the datasets did not include any information about co-products. However, differences between the amount of fruits and vegetables harvested in these scenarios and the final amount available for sale indicates that a portion of the production was unsalable for a variety of reasons. Due to a lack of data on the pathways for these fruits and vegetables and their assumed value, ICF determined that the impacts from any possible co-products from apples, oranges, melons, and tomatoes are outside the scope of this effort. Luske 2010 determined that approximately 10% (by mass) of the bananas produced within the scope of its assessment were unsuitable for international sale and sold to a separate distributor for a much lower price for local distribution. Relative to the price of the bananas destined for international sale, these bananas had approximately 0.3% of the value of the entire yield. Because of the low value and lack of distribution to the United States, ICF deemed that impacts from this co-product were outside of the scope of analysis. The LCI data used to develop included a co-product of potato leaves; however, in the dataset, it was allocated at 0.0% due to its low economic value. Consequently, it was not included in this analysis.

2.7. Dairy Methodology

The LCI data for dairy production used for developing the energy and emission factors in WARM are provided by the Innovation Center for U.S. Dairy, an industry group. The Innovation Center conducted its own LCA for dairy production (Thoma et al. 2010). The study followed the ISO 14040 protocols for LCA and was subject to external review. The Innovation Center's LCA's scope is larger than the scope used to develop the WARM energy and emission factors, covering the cradle-to-grave life-cycle of dairy products including retail storage, consumer use, and disposal. Therefore, ICF removed portions of the unit processes in the LCI data set that were outside the scope of the analysis, such as retail storage, consumer transport, packaging, and consumer use (e.g., cooking and consumer food loss).

The broad category of dairy foods includes a wide variety of products with differing inputs and processing stages. This made it necessary for ICF to develop weighted average energy and emission factors based on the Innovation Center's LCI data that reflect the relative contribution of different products to the total U.S. waste stream. ICF used the USDA Economic Research Service (ERS) loss-adjusted food availability data from 2010 to determine the relative shares of various dairy products within the U.S. waste stream. This dataset includes a variety of more granular product categories than the products addressed in the Innovation Center's LCI data, such as sub-categories of milks and cheeses. In order to make the USDA food loss dataset more consistent with the more general LCI data, ICF consolidated several product categories into more general categories. For example, ICF consolidated a variety of Italian-style cheeses into the Mozzarella category and categorized flavored milk (e.g., chocolate milk) and buttermilk as "generic milk".

In the Innovation Center's LCA, dairy production is linked to several other systems that produce products outside the scope of the WARM food waste source reduction factors, including feed co-products (e.g., dried distillers' grains) and beef. In the data set from the Innovation Center, impacts for most co-products are allocated economically. However, causal allocation is used for both beef based on feed nutrient content and for corn silage based on crop nitrogen requirements determined from reported yield. Causal mass balance is used for different fat-content milks during production (Thoma et al. 2010). Because the Innovation Center's data set already allocated impacts to co-products, ICF did not further modify the data to account for impacts from products outside the scope defined in this memorandum.

ICF analyzed retail transportation of dairy products as a separate unit process and assumed it applied to all dairy products equally. The Innovation Center dataset includes complete LCI data on the retail transportation process for dairy products including energy and emissions from onboard refrigeration equipment to prevent spoilage. This approach differs from the methodology used for estimating retail transport for other materials currently in WARM, which rely on average commodity retail transportation distances provided by the U.S. Census Bureau data. As for dairy production, ICF estimated the energy-derived emissions from transport by calculating the cumulative energy demand within the software. Non-energy emissions in the form of fugitive refrigerants were evaluated with the non-fossil-derived GHG emissions impact assessment method within the software.

3. Results

Using the methodology described in Section 2, ICF developed energy and GHG emissions factors for source reduction of grains, fruits and vegetables, and dairy products. The GHG emissions factors and energy factors are presented in Table 4 and Table 5, respectively, for each category. These factors are integrated into WARM Version 13, available at http://epa.gov/epawaste/conserve/tools/warm/index.html.

Table 4. GHG Emission Factors for Food Waste Categories in WARM

Category	Component	Energy emissions ($MTCO_2e$/tonne)	Non-energy emissions ($MTCO_2e$/tonne)	Total emissions ($MTCO_2e$/tonne)
Grains	Wheat flour	0.27	0.34	0.61
	Corn	0.48	0.20	0.67
	Rice	0.66	0.34	1.00
	Weighted average	0.37	0.31	0.68
Bread	Wheat bread	0.40	0.34	0.74
Fruits and vegetables	Potatoes	0.29	0.06	0.35
	Tomatoes	0.46	0.09	0.55
	Oranges	0.53	0.07	0.59
	Melons	0.32	0.06	0.37
	Apples	0.52	0.02	0.54
	Bananas	0.38	0.12	0.51
	Weighted average	0.41	0.07	0.48
Dairy	Weighted average	0.94	0.98	1.92

Note: $MTCO_2e$ = Metric tons of carbon dioxide equivalent.

Table 5. Energy Factors for Food Waste Categories in WARM

Category	Component	Process and transport energy (GJ/tonne)
Grains	Wheat flour	4.98
	Corn	8.42
	Rice	11.54
	Weighted average	6.54
Bread	Wheat bread	7.57
Fruits and vegetables	Potatoes	4.67
	Tomatoes	7.05
	Oranges	8.01
	Melons	4.75
	Apples	7.98
	Bananas	5.51
	Weighted average	5.90
Dairy	Weighted average	16.61

For grains, wheat has the smallest upstream GHG and energy footprint but constitutes the largest share of the three grains in the waste stream and therefore has the largest impact on the final emission factor. Though the total emissions from rice are significantly higher than both corn and wheat, its smaller share of the overall waste stream reduces its impact on the final weighted emission factor. The largest contributors to the emission factors for all three grains were irrigation, fertilizer production and application, and grain drying—though irrigation was a much lower contributor for wheat production than both corn and rice. The remaining unit processes contributed, on average, less than 10% of the remaining life-cycle impacts.

For bread, roughly one-third of the total energy demand comes from bread baking, leading to an increase in upstream GHG footprint relative to wheat.

For fruits and vegetables, the largest components of the total emission factor are process emissions from cultivation, including on-farm fuel use and upstream production of inputs such as fertilizer. Fertilizer application emissions did not constitute as large a share for these crops as they did for grains, likely due to lower fertilizer application rates for perennial crops. As expected, banana production had the highest life-cycle emissions of all of the fruits and vegetables assessed due to relatively higher fertilizer inputs and a substantial international transport component.

For dairy products, the largest components of the total emission factor are process emissions from dairy agriculture, including both fermentation and fertilizer. Interestingly, one of the largest contributors to energy demand is the production of feed for dairy cattle.

4. Discussion and Conclusion

The results show that the dairy products are both more energy and GHG emissions intensive than grains, bread, and fruits and vegetables. However, the grains and fruits and vegetable factors likely underestimate the GHG emissions associated with grains and fruits and vegetables due to the unavailability of national average LCI data for the processing of those food types into food products. Therefore, in future efforts, ICF will seek to estimate the additional energy and GHG emissions associated with some common processed food types, such as pasta and ketchup.

ICF is currently engaged in the development of similar factors for poultry and red meat. Future efforts will also involve the development of food type-specific end-of-life management factors for all relevant waste pathways in WARM, including composting, landfilling, combustion, and anaerobic digestion.

ICF expects that the streamlined energy and GHG emission factors for food products in WARM will have a broad appeal to parties who manage food waste with an interest in waste reduction. The factors are intended to expand the resolution of food waste modeling in WARM while keeping the tool streamlined, user-friendly, and appropriate for users without a technical background in LCA. EPA expects to apply these factors in a number of existing and future tools and programs. For example, the WARM factors are expected to be used as part of EPA's Food Recovery Challenge, a voluntary program for commercial, governmental, and non-commercial organizations. Food Recovery Challenge participants will use the source reduction factors to estimate the GHG emissions savings from food waste reduction efforts over the past calendar year.

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Designing healthy, climate friendly and affordable school lunches

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ABSTRACT

It is well-known that the feeding of students is related to their health and well-being, contributing to a reduction in the risk of chronic diseases in adulthood. However, students' parents also take the cost of school lunches into account. In addition, we should not forget that the food sector contributes to 15-30% of the total greenhouse gas emissions and, as such, food choices can have a notable influence on climate change. Although these three aspects, health, environment and cost, need to be borne in mind in order to build sustainable lunches, they are not necessarily convergent. In this context, this study aims to develop a model with which building a sustainable school diet while taking nutritional, environmental and economic aspects into account. Interval goal programming can be very useful as a means of addressing this problem. By establishing two goals, an economic one and a carbon footprint (CFP) one, positive deviations can be minimized while keeping the recommended daily intake (RDI) fractions (macronutrients and micronutrients) and energy in the expected range.

Keywords: goal programming, nutrition, carbon footprint, sustainability, school lunch

1. Introduction

It is well-known that the feeding of students is related to their health and well-being, contributing to a reduction in the risk of chronic diseases in adulthood. However, students' parents also take the cost of school lunches into account. In addition, we should not forget that the food sector contributes to 15-30% of the total greenhouse gas emissions and, as such, food choices can have a notable influence on climate change. These three aspects, health, climate change and cost, need to be borne in mind in order to build sustainable lunches. The FAO definition of a sustainable diet reads "those diets with low environmental impacts which contribute to food and nutrition security and to healthy life for present and future generations. Sustainable diets are protective and respectful of biodiversity and ecosystems, culturally acceptable, accessible, economically fair and affordable; nutritionally adequate, safe and healthy; while optimizing natural and human resources" (FAO, 2010). However, these aspects are not necessarily convergent. In this context, this study aims to develop a model with which to build a school diet taking nutritional, climate change and economic aspects into account. The contribution of food to global warming has been assessed via carbon footprinting, a method for the quantification of greenhouse gas emissions emitted throughout the life cycle of a product (BSI, 2011). Although the carbon footprint (CFP) represents only one part of the environmental impacts of food products, the relationship between food consumption patterns and climate change is a subject of great concern (Carlsson-Kanyama and González, 2009). In spite of the limitations of assessing only one impact, CFP presents several advantages that have made it a commonly used indicator for the eco-labeling of food products. These advantages include a connection to global warming, which is internationally acknowledged as an important environmental concern. Furthermore, a major advantage is the reduced complexity when computing and interpreting the results (Weidema et al., 2008; Heller et al., 2013), which is related to the global character of climate change. Other impact categories, such as land use and toxicity, also relevant in food life cycles, are associated with regional impacts, making data sets less applicable depending on the region and adding uncertainty to the calculations.

Interval goal programming can be very useful as a means of designing diets which take the nutritional, climate change and economic aspects into account. By establishing two goals, an economic one and a CFP one, positive deviations can be minimized while keeping the recommended daily intake (RDI) fractions (macronutrients and micronutrients) and energy content in the expected range.

A case study is used to test the proposed model. From 20 starters, 20 main courses and 4 desserts, combinations were made to obtain 1,600 lunch menus. The criteria established for the composition of the menus tried to ensure the nutritional quality and familiarity of the foods and CFP data availability. The objective of the model is to design a lunch for an elementary school student for four school weeks, bearing in mind not only global warming, but also nutritional and economic aspects. An optimizing technique, specifically Goal Programming (GP), is used to choose those menus that fulfill the requirements. In the context of decision making, to optimize means to find the decision which gives the best possible value of some measures from amongst the set of possible decisions (Jones and Tamiz, 2010). Other optimization techniques, such as Linear Programming, have been previously used to assess whether a reduction in greenhouse gas emissions can be achieved while meeting dietary health requirements (Macdiarmid, 2012).

2. Methods

In order to make decisions which take several, non-convergent criteria into account implies the application of a weighting method, namely Multicriteria Decision Making Methods (MCDM). There is a wide range of MCDM, but in order to design sustainable menus we do need a flexible method that enables different kinds of goals to be integrated; i.e. to minimize cost and CFP levels, to ensure that the energy content is between a minimum and maximum and also to make sure that the kilocalories shared out among the macronutrients are set in the right intervals (50-55% carbohydrates, 12-15% proteins and 30-35% fat). Simultaneously, some micronutrients should be encouraged (vitamin A, vitamin C, vitamin D, calcium, iron, magnesium, …) while others should be limited (sodium, saturated fat,…) (Drewnowski, 2009). Moreover, the importance of each criterion can be different: e.g. we want to be sure that the amount of calcium is above 1000 mg for a 2,000-kcal diet on a daily basis (strong condition) and we would like the carbon footprint to be around a certain level (weak condition). Goal programming allows all these conditions to be addressed when designing diets.

2.1. Case study

A case study is used to test the proposed model. From 20 starters, 20 main courses and 4 desserts (Table 1), combinations were made to obtain up to 1,600 lunch menus that were assessed. The criteria for choosing the dishes tried to ensure the nutritional quality and familiarity of foods and the availability of CFP information.

The CFP of each dish was assessed based on literature about life cycle assessment and CFP studies into food products (around 50 literature sources). The PAS 2050:2011 guidelines (BSI, 2011) were used to analyze all CFP data sources in terms of their system boundaries, completeness and appropriateness. Where no data was available, the impact of packaging was assumed to be negligible (Jungbluth et al., 2000). Since some literature sources consider only primary production, when energy from the processing step was missing, adaptations were made following Sanjuan et al. (2014). The same procedure applied to the use phase, which was either part of the literature source or was included by using data from Carlsson-Kanyama and Faist (2000). The transport processes were either included in the literature sources or added manually, taking into account both the distance from the food origin to the consumption point (Valencia, Spain) and also the means of transport from Ecoinvent v 2.0. As regards transportation, the distance from the retailer to the consumer was not taken into account according to PAS 2050:2011. Cooling during transportation was added if not previously included. The operation of premises and the storage of products could not be included manually due to lack of data. However, where it was included in the literature, it was taken into account.

The cost was calculated from the market prices of the raw foods. The source for food prices was the web page of the three biggest supermarket chains in Spain. Only the price of raw materials has been taken into account, neither labor costs nor other direct costs have been computed. Additionally, the caloric content together with the macronutrient (protein, fat and carbohydrate) and micronutrient content (namely fibre, calcium, iron, potassium, magnesium, vitamin A, vitamin C, vitamin D, cholesterol and sodium) of each food were obtained from databases (USDA, 2011; software dietowin® 7.3).

In order to fix the threshold values of nutrients to be used as restrictions in the program, the reference daily values were obtained from the literature (Cuervo et al., 2009; Drewnowski, 2012; FAO, 2012). Fibre, calcium, iron, potassium, magnesium, vitamin A, vitamin C, vitamin D are considered as nutrients to be encouraged, while it was considered that two nutrients should be limited, specifically cholesterol and sodium. Table 2 shows the bounds used as a minimum, a maximum or as an interval.

Table 1. Dishes from which the daily menus were built

	Dish	CFP kg CO2-eq	PRICE €	CALORIES kcal	PROTEIN kcal	FAT kcal	CH kcal	Fibre g	Calcium mg
S01	Peas with ham	0.46	1.28	437.49	84.10	143.67	219.40	18.64	97.64
S02	Green salad with lentils, cheese and ham	0.13	1.10	389.58	121.04	200.37	72.56	4.97	718.07
S03	Rice salad with apple	0.13	0.29	74.92	5.98	1.49	66.88	1.22	13.56
S04	Lentils with potatoes	0.17	0.32	472.55	105.67	44.55	338.46	14.76	68.24
S05	Macaroni with three cheeses	0.47	0.89	330.23	83.38	189.25	56.02	0.80	563.92
S06	Potato salad with olive oil	0.91	0.78	387.05	61.48	251.90	76.08	4.33	50.22
S07	Onion soup	0.41	0.44	300.07	46.95	49.09	219.66	4.26	219.64
S08	Pasta salad	0.17	0.31	124.80	14.84	35.01	74.25	2.41	25.92
S09	Boiled spinach with olive oil	0.28	1.66	129.67	66.98	13.19	84.57	13.53	766.74
S10	Mussels with raw onion and tomato	1.85	0.41	238.03	117.05	66.70	46.68	1.01	50.90
S11	Sautéed green beans	0.48	1.91	315.26	24.70	237.09	56.38	7.54	110.10
S12	Salad with raisins	0.14	1.03	262.90	47.89	144.62	74.97	4.20	446.20
S13	Rice with tuna and onion	0.79	0.59	232.05	47.39	92.27	90.05	0.50	23.10
S14	Carrot purée	0.23	0.33	191.22	24.85	35.96	139.90	8.16	199.06
S15	Spaghetti bolognese	1.03	1.17	373.59	78.19	98.65	192.34	3.10	196.96
S16	Boiled potatoes and green beans with olive oil	0.57	1.44	282.70	33.63	51.69	203.11	10.29	102.84
S17	Minestrone soup	0.63	1.38	271.32	54.62	70.02	157.80	8.88	350.58
S18	Spinach and cheese quiche	0.28	0.51	336.27	60.36	172.19	106.54	2.73	388.23
S19	Boiled broccoli with olive oil and lemon	1.68	1.76	402.05	31.86	292.66	107.29	17.27	691.93
S20	Pasta with onion	0.15	0.12	128.87	13.54	42.06	72.71	1.19	6.93
M01	Grilled hake	1.29	1.79	162.51	84.27	74.00	4.24	0.00	37.10
M02	Pork chop and boiled potatoes	0.33	0.62	210.42	64.40	109.85	32.79	0.82	20.45
M03	Fried horse mackerel	0.60	9.72	525.08	279.51	225.56	0.00	0.00	38.51
M04	Grilled chicken	1.33	0.47	332.74	129.29	194.65	0.27	0.00	17.54
M05	Sole Meunière	1.72	2.41	269.77	118.92	100.58	49.50	0.34	82.17
M06	Trout with tomato sauce	0.48	0.59	212.95	60.12	136.78	13.59	2.58	31.94
M07	Grilled burger	5.41	5.83	299.93	187.96	99.94	0.00	0.00	24.99
M08	Horse mackerel with tomato and onion	0.77	7.40	475.20	209.32	226.92	26.58	4.18	62.35
M09	Chicken à'lorange	1.76	1.12	470.33	117.06	187.18	172.02	7.39	132.51
M10	Battered hake	2.02	2.66	361.97	151.64	139.10	68.29	0.54	81.32
M11	Tuna in papillote	4.05	2.78	284.74	213.42	12.12	48.32	3.65	53.14
M12	Veal and ham meatballs	3.80	4.10	337.10	144.81	110.20	73.73	0.67	23.11
M13	Cod Vizcaya style	0.79	1.42	367.95	132.93	94.44	136.87	4.05	45.59
M14	Roast chicken with vegetables	1.25	0.69	408.78	126.30	193.16	86.60	6.06	53.69
M15	Trout with onion and tomato sauce	0.47	0.70	226.41	96.40	109.98	15.80	0.99	39.87
M16	Fried monkfish	2.06	1.49	317.30	126.78	183.37	1.09	0.02	18.41
M17	Grilled beefsteak with onion	4.54	4.77	328.68	156.70	122.79	43.39	1.59	44.98
M18	Cod with vegetables	0.65	1.20	264.72	120.47	127.58	12.11	0.43	25.63
M19	Pork with green beans, tomato and onion	0.48	0.93	197.95	55.10	112.28	30.28	3.34	50.73
M20	Grilled chicken with baked potatoes	1.84	0.70	471.13	183.40	219.27	57.99	1.00	27.51
D01	Yogurt	0.18	0.14	66.36	15.36	33.48	18.72	0.00	150.00
D02	Orange	0.04	0.21	38.33	3.50	0.00	37.67	2.19	39.42
D03	Banana	0.18	0.27	82.17	4.75	2.67	79.20	3.37	8.91
D04	Apple	0.08	0.36	57.96	1.51	0.00	60.48	2.52	7.56

Table 1 (continued).

	Dish	Iron	Potassium	Magnesium	Vit C	Vit A	Vit E	Cholesterol	Sodium
		mg	mg	mg	mg	mg	mg	mg	mg
S01	Peas with ham	5.57	988.47	135.14	48.60	132.50	2.43	6.78	1009.98
S02	Green salad with lentils, cheese and ham	3.55	565.65	67.50	9.22	752.09	0.70	75.28	614.81
S03	Rice salad with apple	0.93	115.86	10.33	6.60	8.78	0.20	0.00	3.00
S04	Lentils with potatoes	7.59	1238.84	108.60	14.32	10.00	0.17	0.00	380.73
S05	Macaroni with three cheeses	0.65	133.07	34.76	0.00	193.49	0.24	65.71	494.02
S06	Potato salad with olive oil	1.62	505.13	42.09	10.26	457.39	4.39	58.04	550.49
S07	Onion soup	1.64	425.17	46.21	10.92	39.83	0.09	15.00	893.56
S08	Pasta salad	0.75	228.18	17.88	12.04	401.94	0.80	0.00	24.20
S09	Boiled spinach with olive oil	20.13	2627.20	490.49	55.25	2954.19	11.73	0.00	1725.16
S10	Mussels with raw onion and tomato	8.52	530.62	52.24	34.10	161.67	0.78	66.97	444.00
S11	Sautéed green beans	2.51	771.61	68.54	64.63	357.63	2.56	0.00	262.82
S12	Salad with raisins	1.51	550.48	42.71	17.29	516.56	1.12	35.01	199.05
S13	Rice with tuna and onion	1.65	169.93	23.10	1.07	24.64	2.59	26.70	221.58
S14	Carrot purée	1.10	868.14	46.56	15.20	1946.00	2.41	13.39	943.63
S15	Spaghetti bolognese	2.32	220.20	44.36	1.70	43.33	0.73	40.08	363.36
S16	Boiled potatoes and green beans with olive oil	2.59	1262.50	96.37	66.89	60.82	0.28	0.00	485.10
S17	Minestrone soup	3.19	1091.74	73.07	69.88	1286.82	2.66	22.70	1226.45
S18	Spinach and cheese quiche	4.87	472.38	90.95	8.02	535.72	3.20	100.79	381.96
S19	Boiled broccoli with olive oil and lemon	4.01	1844.45	126.53	206.85	566.09	4.72	0.00	48.61
S20	Pasta with onion	0.76	47.17	11.44	0.72	0.00	0.63	0.00	105.22
M01	Grilled hake	1.08	480.98	30.48	0.00	0.00	0.23	88.78	98.05
M02	Pork chop and boiled potatoes	1.13	405.96	24.73	3.67	1.91	0.31	54.06	154.70
M03	Fried horse mackerel	2.22	1641.03	112.56	4.74	97.75	0.32	216.24	195.50
M04	Grilled chicken	2.19	275.20	26.98	0.54	241.47	0.20	126.81	95.78
M05	Sole Meunière	2.01	681.50	45.82	0.46	64.43	0.16	134.24	141.46
M06	Trout with tomato sauce	0.67	533.02	29.63	12.45	137.34	1.96	41.48	58.66
M07	Grilled burger	1.62	651.64	48.20	0.00	0.00	0.82	185.67	151.75
M08	Horse mackerel with tomato and onion	2.62	1683.96	103.96	25.54	178.63	0.25	155.95	155.85
M09	Chicken à'lorange	3.76	1317.04	78.63	98.79	1861.22	2.30	94.07	586.59
M10	Battered hake	3.23	776.56	53.58	0.00	87.30	1.11	288.06	224.60
M11	Tuna in papillote	2.65	1395.86	95.44	26.06	1004.06	1.76	82.79	263.38
M12	Veal and ham meatballs	2.19	500.53	40.78	0.06	0.18	1.09	132.56	157.14
M13	Cod Vizcaya style	2.43	657.75	76.80	11.58	94.46	1.23	67.77	313.78
M14	Roast chicken with vegetables	3.16	658.36	60.27	14.17	771.59	1.66	106.42	502.43
M15	Trout with onion and tomato sauce	0.71	614.81	36.67	16.08	141.19	3.99	68.55	119.72
M16	Fried monkfish	0.81	896.99	47.23	7.84	24.21	2.45	54.56	39.73
M17	Grilled beefsteak with onion	1.59	710.25	51.09	5.90	0.00	0.91	148.70	392.75
M18	Cod with vegetables	0.79	380.13	58.60	6.46	18.40	2.92	71.50	173.46
M19	Pork with green beans, tomato and onion	1.58	539.53	36.96	20.78	51.40	0.62	40.44	129.04
M20	Grilled chicken with baked potatoes	3.23	640.59	53.89	9.30	332.75	0.03	174.74	293.08
D01	Yogurt	0.12	186.00	16.80	0.48	13.20	0.04	8.40	64.80
D02	Orange	0.33	219.00	13.14	54.75	53.66	0.22	0.00	3.29
D03	Banana	0.59	346.50	37.62	9.90	17.82	0.20	0.00	0.99
D04	Apple	0.50	151.20	6.30	12.60	5.04	0.25	0.00	2.52

Table 2. Threshold values of the nutrients (expressed on daily lunch basis)

Nutrient	Minimum or lower bound	Maximum or upper bound
Caloric content (kcal)	600	800
% fat calories	30	35
% protein calories	12	15
% carbohydrate calories	50	55
Fibre (g)	7.5 g	
Calcium (mg)	240	
Iron (mg)	2.70	
Potassium (mg)	600	
Magnesium (mg)	75	
Vitamin A(μg)	120	
Vitamin C (mg)	16.50	
Vitamin E (mg)	2.40	
Cholesterol (mg)		116
Sodium (mg)		720

2.2. Goal program

Each goal program needs an objective function to optimize, which usually consists of minimizing the unwanted deviations of some goals. In our case study, these deviational variables come from the nutrient content, CFP and price constraints. Negative deviations are set from the nutrients to be encouraged and positive deviations from those to be limited. Caloric content and macronutrient content make up interval goals (lower bound-upper bound) and, therefore, negative deviations have to be minimized from the lower bounds and positive deviations from the upper ones.

The objective function is made up of four main addends, the first one includes positive deviations from the CFP and price goals, the second one includes the daily negative deviations from the lower bound of caloric content (600 kcal) and the daily positive deviations from the upper bound of caloric content (800 kcal). The third addend is referred to the macronutrient composition as the share of the energy content and includes a negative deviation from the lower bound and a positive deviation from the upper bound for each macronutrient and day. Other nutrients to be encouraged and limited build the fourth addend, specifically negative deviations from the lower bound of the nutrients to be encouraged and positive deviations from the upper bound of the nutrients to be limited.

A weight of 25% has been applied to each addend that represents the relative importance of the addend to the decision maker. At the same time, each deviational variable has been divided by a normalization factor that scales the deviations so they can be compared in the same units. The normalization factor is taken from the bound of the respective aspect (CFP, price, nutrients…). Since providing nutrients is the main function of food, 75% of the weight has been allocated to nutrient-related variables and the remaining 25% jointly to the price and CFP.

The decision variables are: (20 starters +20 main dishes + 4 desserts) x 20 days = 880 variables

Xs_{ij}: variable of starter *j* on day *i*

Xs_{ij}: variable of main dish *j* on day *i*

Xd_{ij}: variables of dessert *j* on day *i*

All of them are binary variables; they only can take the values 0 or 1. For example, if Xsij=1, it means that starter j is part of the lunch on day i, otherwise starter j is not part of the lunch on day i.

There are 6 groups of constraints in the goal program:

- Budget constraint: 1 constraint for the whole planning period
- CFP constraint: 1 constraint for the whole planning period
- Energy content constraints: (20 days x 2 levels [lower and upper]) = 40 constraints
- Macronutrient constraints: (20 days x 2 levels x 3 macronutrients) = 120 constraints
- Nutrients-to-encourage constraints: (20 days x 8 nutrients) = 160 constraints
- Nutrients-to-limit constraints: (20 days x 2 nutrients) = 40 constraints

There are also some other constraints related to the composition of the menus: the starter and the main dish can not be repeated more than twice in the planning period and the desserts can not be repeated more than five times. Since weights are included in the objective function and some constraints are defined as intervals, the program to be built is a mixture between weighted goal programming and interval goal programming.

Bearing all these features in mind, the goal program can be written this way, *i* being: number of day (1 to 20); *j:* number of dish (1 to 20 for starters and main dish, 1 to 4 for desserts); and *k:* number of nutrient (1 to 8 for nutrients to be encouraged and 1 to 2 for nutrients to be limited).

$$MIN\left[p_{budget} / RHS_{budget} + p_{CFP} / RHS_{CFP}\right]\cdot w_1 + \left[\sum_i^{20}\left(n\_cal_i^L / RHS_{cal}^U + p\_cal_i^U / RHS_{cal}^U\right)\right]\cdot w_2 +$$

$$+\left[\sum_i^{20}\begin{pmatrix} n\_fat_i^L / RHS_{fat}^L + n\_prot_i^L / RHS_{prot}^L + n\_CH_i^L / RHS_{CH}^L + p\_fat_i^U / RHS_{fat}^U + \\ + p\_prot_i^U / RHS_{prot}^U + p\_CH_i^U / RHS_{CH}^U \end{pmatrix}\right]\cdot w_3$$

$$+\left[\sum_i^{20}\begin{pmatrix} n\_water_i + n\_fiber_i + n\_calcium_i + n\_iron_i + n\_pota_i + n\_magne_i + n\_vitA_i + \\ + n\_vitC_i + n\_vitE_i + p\_cholest_i + p\_sodium_i \end{pmatrix}\right]\cdot w_4$$

$s.t.$

Budget constraint :

$$\sum_i^{20}\left[\sum_j^{20} Xs_{ij}\cdot price\_s_j + Xm\cdot price\_m_j + \sum_j^{4} Xd_{ij}\cdot price\_d_j\right] - p_{budget} + n_{budget} = RHS_{budget}$$

CFP constraint :

$$\sum_i^{20}\left[\sum_j^{20} Xs_{ij}\cdot CFP\_s_j + Xm\cdot CFP\_m_j + \sum_j^{4} Xd_{ij}\cdot CFP\_d_j\right] - p + n = RHS_{CFP}$$

Caloric content constraints :

$$\sum_j^{20} Xs_{ij}\cdot cal\_s_j + Xm_{ij}\cdot cal\_m_j + \sum_j^{4} Xd_{ij}\cdot cal\_d_j - p_{cal\_i}^L + n_{cal\_i}^L = 600 \ \forall i = 1..20 days$$

$$\sum_j^{20} Xs_{ij}\cdot cal\_s_j + Xm_{ij}\cdot cal\_m_j + \sum_j^{4} Xd_{ij}\cdot cal\_d_j - p_{cal\_i}^U + n_{cal\_i}^U = 800 \ \forall i = 1..20 days$$

Macronutrient constraints :

$$\left(\sum_j^{20} Xs_{ij}\cdot prot\_s_j + Xm_{ij}\cdot prot\_m_j + \sum_j^{4} Xd_{ij}\cdot prot\_d_j - p_{CH\_i}^L + n_{CH\_i}^L\right)/600 = 0.12 \qquad \forall i = 1..20 days$$

$$\left(\sum_j^{20} Xs_{ij}\cdot prot\_s_j + Xm_{ij}\cdot prot\_m_j + \sum_j^{4} Xd_{ij}\cdot prot\_d_j - p_{CH\_i}^U + n_{CH\_i}^U\right)/800 = 0.15 \qquad \forall i = 1..20 days$$

$$\left(\sum_j^{20} Xs_{ij}\cdot fat\_s_j + Xm_{ij}\cdot fat\_m_j + \sum_j^{4} Xd_{ij}\cdot fat\_d_j - p_{fat\_i}^L + n_{fat\_i}^L\right)/600 = 0.30 \ \forall i = 1..20 days$$

$$\left(\sum_j^{20} Xs_{ij}\cdot fat\_s_j + Xm_{ij}\cdot fat\_m_j + \sum_j^{4} Xd_{ij}\cdot fat\_d_j - p_{fat\_i}^U + n_{fat\_i}^U\right)/800 = 0.35 \ \forall i = 1..20 days$$

$$\left(\sum_j^{20} Xs_{ij}\cdot CH\_s_j + Xm_{ij}\cdot CH\_m_j + \sum_j^{4} Xd_{ij}\cdot CH\_d_j - p_{CH\_i}^L + n_{CH\_i}^L\right)/600 = 0.50 \qquad \forall i = 1..20 days$$

$$\left(\sum_j^{20} Xs_{ij}\cdot CH\_s_j + Xm_{ij}\cdot CH\_m_j + \sum_j^{4} Xd_{ij}\cdot CH\_d_j - p_{CH\_i}^U + n_{CH\_i}^U\right)/800 = 0.55 \qquad \forall i = 1..20 days$$

Nutrients-to-encourage constraints:

$$\left(\sum_{j}^{20} Xs_{ij} \cdot encou_{k}\_s_j + Xm_{ij} \cdot encou_{k}\_m_j + \sum_{j}^{4} Xd_{ij} \cdot encou_{k}\_d_j - p^{U}_{CH\_i} + n^{U}_{CH\_i}\right) \leq RHS_{encour\_i} \qquad \forall i = 1..20 days$$

$\forall k =$ fiber, calcium, iron, potasium, magnessium, vitA, vitC, vitE

Nutrients-to-limit constraints:

$$\left(\sum_{j}^{20} Xs_{ij} \cdot \mathrm{limit}_{k}\_s_j + Xm_{ij} \cdot \mathrm{limit}_{k}\_m_j + \sum_{j}^{4} Xd_{ij} \cdot \mathrm{limit}_{k}\_d_j - p^{U}_{CH\_i} + n^{U}_{CH\_i}\right) = RHS_{\mathrm{lim}it\_i} \qquad \forall i = 1..20 days$$

$\forall k =$ cholesterol, sodium

Decision variables constraints:

$$\sum_{i}^{20} \mathrm{Xs}_{\mathrm{ij}} \leq 2 \qquad \forall j = 1..20 \text{ starters}$$

$$\sum_{i}^{20} \mathrm{Xm}_{\mathrm{ij}} \leq 2 \qquad \forall j = 1..20 \text{ main dishes}$$

$$\sum_{i}^{20} \mathrm{Xd}_{\mathrm{ij}} \leq 5 \qquad \forall j = 1..4 \text{ desserts}$$

$$\sum_{i}^{20}\sum_{j}^{20} Xs_{ij} \geq 20 \qquad \sum_{i}^{20}\sum_{j}^{20} Xm_{ij} \geq 20 \qquad \sum_{i}^{20}\sum_{j}^{20} Xd_{ij} \geq 20$$

In order to set the goals for the price and the CFP for the planning period, the total price of the average menu for 20 days (20-day budget: 74.01 €) and the CFP (48.99 kg CO2 eq) have been calculated. The average menu for 20 days comes from table 1 by repeating the four desserts five times, each row representing the lunch for one day. To carry out further analysis, the percentile values of CFP and the price of the 1,600 combination menus have also been calculated. The program has been written in Lingo modeling language and solved by means of the LINGO software (ver.6.0).

3. Results and discussion

After running the program, the CFP for the planning period equals 36.96 kg CO_2 eq (-25% of the CFP goal) while the price equals 48.90 € (-34% of the price goal). The energy content, fat content, nutrients to be encouraged and nutrients to be limited exhibit zero deviation but the percentage of protein and carbohydrates are beyond the range. The solutions of the program (starter, S, main dish, M, and dessert, D) for each day show that some dishes are not part of the solution while others appear twice in the planning period. For example, the main dish of tuna (M11) is not included in the planning because of its high contribution to the CFP, while the main dish of horse mackerel (M03) is not included because of its price.

Figure 1 has been built for the purposes of comparing the optimal menu of the goal program with the average menu and the 1,600 combination menus. By considering the 1,600 combination menus and extending each menu over the 20-day period, the CFP has been calculated and plotted for 20 days as has the 20-day budget for each menu (single menu x 20). The CFP and budget of the average 20-day menu has also been plotted. Figure 1 clearly shows how the 20-day menu obtained by means of the goal program outperforms the average menu in both economic and CFP terms.

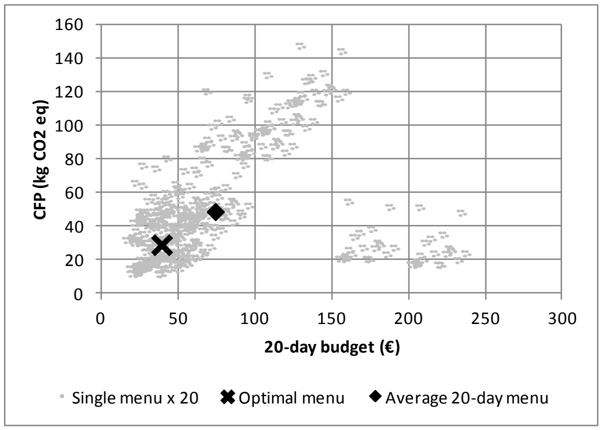

Figure 1. Comparison between optimal menu, 1,600 menus and average 20-day menu for a 20-day period

From the nutritional point of view, the energy content of both the average 20-day menu and the optimal menu is located in the recommended interval. The energy content distribution among macronutrients is unbalanced in both the average 20-day menu and the optimal menu, although the optimal menu is closer to the recommendations (Table 3). As far as the nutrients to be encouraged are concerned, both menus comply with the reference values. In the case of nutrients to be limited, whereas the cholesterol and sodium levels are exceeded in the average menu, the optimal menu meets both goals (Table 4).

Table 3. Energy content of macronutrients

% Kcal	Protein	Fat	Carbohydrate
Lower reference value	12%	30%	50%
Upper reference value	15%	35%	55%
Average menu	25%	35%	40%
Optimal menu	30%	40%	29%

Table 4. Relative deviation of nutrients

Nutrients	Average menu	Optimal menu
Nutrients to be encouraged		
FIBRE	+39%	+26%
CALCIUM	+46%	+15%
IRON	+128%	+61%
POTASSIUM	+188%	+121%
MAGNESIUM	+107%	+53%
VIT C	+296%	+218%
VIT A	+567%	+433%
VIT E	+46%	+9%
Nutrients to be limited		
CHOLESTEROL	+25%	-10%
SODIUM	+4%	-9%

The goal program can be tweaked in order to analyze specific questions. The influence of the budget on the CFP can be tested by taking economic and CFP deviations out of the objective function and forcing the budget to be below specific values: 20th budget percentile, 30th budget percentile... This way the program has been solved for each budget percentile. Figure 2 shows the evolution of the CFP according to the limitations in the lunch budget. As can be observed, a small budget does not lead to a higher CFP, quite the contrary, lower budgets show a lower CFP and, from the 60th percentile, the CFP first drops and then remains steady. From the nutritional point of view, lower budgets exhibit a similar level of deviation from the nutritional goals to the higher budgets.

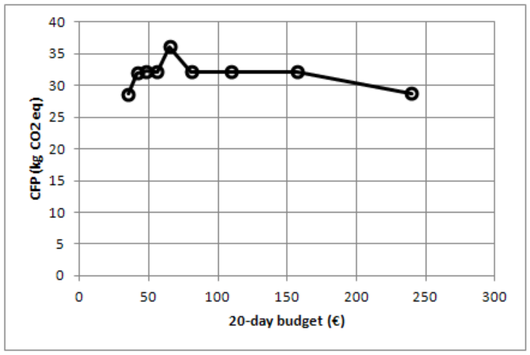

Figure 2. CFP as a function of limited budget

The use of binary variables implies an integer linear program whose solver algorithms are not as efficient as the simplex algorithm used in linear programming, which means that the solving takes longer. Nevertheless, the process can be speeded up dramatically by adjusting some of the parameters of the solver algorithms and solving by first taking only a part of the decision variables as an integer and keeping them as a starting point for a second solving.

This study demonstrates an approach to the design of sustainable diets for schools. Sustainability is a broad, complex and controversial concept. While the FAO (2010) definition of sustainable diets is clear and comprehensive, it does not provide the way to attain that goal. As Garnett (2014) states, the FAO definition suggests that these multiple dimensions are synergistic, when inevitably there will be tradeoffs. Another issue of the definition of sustainable diets is the lack of consistency; stakeholders can choose different parameters to measure each dimension and can assign weights to these dimensions differently. In the present study, only the CFP is used as a means of measuring the environmental performance. The inclusion of more environmental impact categories would have led to a different environmental result. On the other hand, social and ethical aspects, such as labor conditions, the farmer’s livelihood or animal welfare, have not been taken into account. From the economic point of view, the model focuses on the consumer (affordability) and leaves out the producer (economic fairness). It must be pointed out that goal programming models allow more environmental, economic or social aspects to be included.

4. Conclusion

Goal programming constitutes a suitable tool for designing economically, environmentally and nutritionally sustainable diets. Its flexibility enables specific issues to be studied, such as how economic restrictions are linked to CFP, or other environmental impacts. Specific and very realistic constraints can be included; for example, weak restrictions can be considered on a daily basis, e.g. the sodium content might be around 720 mg, while, at the same time, taking into account strong restrictions over the whole planning period e.g. the sodium content has to be lower than 14,400 mg (720 mg x 20 days). Moreover, weak restrictions are more suitable for taking the micronutrient content into account, since the nutritional recommendations vary depending on the source. New constraints, such as including nutritional requirements for a specific population (e.g. allergy sufferers, people suffering from coeliac disease or religious groups) can also be added to the goal program. In the same way, other nutritional recommendations could be incorporated into the program, such as the quantity of monounsaturated fatty acids, saturated fat, monounsaturated fat, zinc or folate.

The case study shows how an optimal design of the school lunch menus makes it possible to reduce the carbon footprint at affordable prices. It also shows that lower food budgets are neither linked to a higher carbon footprint nor to a worse nutritional performance. Comparing the 20-day optimal menu, designed by means of the goal program, with the 20-day average menu, the former not only reduces the CFP by 25% and the food budget by 34%, but also improves the balance among macronutrients, meets the requirements in terms of the nutrients to encourage and is below the threshold level in the case of the nutrients to limit.

The lack of precision in the definition of a sustainable diet makes the task of designing them difficult, and also hampers the measurement of diet sustainability.

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Environmental Impact Evaluation of Beef Production in Veracruz Using Life Cycle Assessment

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ABSTRACT

Life-Cycle Assessment (LCA) was utilized to compare the potential for climate change, acidification and human toxicity associated with the production of boneless beef sold to the consumer through two chains of production in Veracruz, Mexico. An intensive method based on feedlot finishing and highly technified transformation and marketing procedures was analyzed and compared to an extensive livestock-farming model up to the finishing period and adopting low technology transformation and marketing procedures. The potential impacts on every category studied were greater in the intensive chain. More methane from enteric fermentation was generated in the extensive chain due to lower quality feed compared to feedlot finishing, however the latter is associated with greater impact within the intensive chain because of production and processing of the feed raw material.

Keywords: LCA, beef , climate change, supply chain, environmental.

1. Introduction

Beef production worldwide has increased more than threefold between 1970 and 2009 (FAO 2012). The beef production supply chain has a significant impact on the environment, at every level of production (Steinfeld et al., 2006) such as the degradation of natural resources as a result of cattle feeding, land use in primary production, fossil fuel consumption, water usage and greenhouse gas emissions.

Life-Cycle Assessment has been applied in different world locations to identify the beef production environmental impact and define key points for its environmental sustainability improvement (de Boer et al., 2010). A Life-Cycle perspective provides also a useful framework to study the links between social needs, natural and economic processes related to food production, and their environmental consequences (Heller and Keoleian, 2003). In Mexico a full study to calculate the impact of the beef production chain on the environment has yet to be carried out. Therefore in this paper we used the Life-Cycle Assessment approach to evaluate two beef production chains in the state of Veracruz, Mexico. A production system involving animal growing on pasture and successively fattened in feedlots where modern technologies are adopted in the transformation and commercialization stages is analyzed, has been compared to another production system based on livestock exclusively bred in pastures and adopting low-level technologies in the transformation and commercialization stages. The objective of this study was to quantify and compare the two systems environmental impact related to both the beef production and the other stages in their supply chains and to identify windows of opportunity for improvement in their sustainability.

2. Methods

The Life-Cycle Assessment (LCA) was carried out according to the ISO 14040 standard. The aim was to compare potentials for climate change, acidification, and human toxicity associated with the two beef production chains above described, characteristic of the North-Central region of the state of Veracruz, Mexico. They will be named *intensive production chain* and *extensive production chain*.

2.1. Goal and Scope Definition

2.1.1. System boundary

The LCA covered the stages from the production of fodder and grain used as cattle feed up to the final consumer. This included manure-handling impact, manure use as farmland fertilizer, transport between the different production stages including transport to the consumer's home, the use of fertilizers, energy, organic residue disposal and packaging material in the transformation and commercialization procedures.

2.1.2. Functional Unit

The functional unit in this LCA is 1 kg of boneless, skinless beef without fat, produced in Veracruz for processing and consumption.

2.1.3. System Description

The study scenarios were put together based on literature review, national and state-level statistics and data collected from producers, beef association representatives, researchers, academics and extension specialists. The production chains studied are made up of five process units: growth and development (G & D), pre-fattening (PrF), fattening (Fat), transformation (Tr), marketing (Market). In the intensive chain G & D and pre-fattening livestock are kept on pasture, fattening is carried out in feedlot, transformation in highly technified slaughterhouses and marketing in supermarkets. In the extensive chain G & D, pre-fattening and fattening are on pasture, transformation takes place in slaughterhouses of low technology and finally, marketing is done by local butchers. The life-cycle inventory is based on average data from 2011 to 2013 for each life cycle, all-representative of Mexico. The production processes of each supply chain, and related parameters are described below:

2.1.4. Growth and Development

The growth and development process is the same in both production chains. Cattle are a mix of Brown Swiss-Zebu and Holstein-Zebu in different proportions. Feed is pasture based with a supplement of 14% raw protein. The model adopted is representative of the North-Central region of Veracruz and consists of 72 production cows, 20 calves and 2 bulls. Fertility in the reproductive cattle is 58.5% and mortality rate is 2%. Calves at birth weight 37 kg and are weaned at 167 days, after that they are pasture fed with a supplement of 14% raw protein feed. 93 hectares of prairie are used, and fertilized with ammonium sulphate (150 kg/ha/year). 44 calves weighing 225 kg and 12 months of age are sold annually. Cattle are transported 10 km to the pre-fattening sites.

2.1.5. Pre-fattening

In the intensive chain, livestock enter the pre-fattening stage weighing 225 kg. They are fed on pasture and 76 days before they are sold they are also given 1 kg/head of raw protein supplement at 15 %. This stage lasts 190 days. Per year 270 head are sold for fattening, each weighing 336 kg. The livestock is transported 10 km.

In the extensive chain pre-fattening takes place on pasture with no supplements. Cattle are received weighing 225 kg and spend 243 days until they weigh 348 kg. The area used measures 240 ha. Per year 362 head go on to fattening. Pre-fattening and fattening are carried out in the same place therefore no transport is necessary in these stages.

2.1.6. Fattening

In the intensive chain fattening is carried out in pens. Animals start this stage weighing 333 kg. Feed is composed of maize, sorghum, and soybean meal. Distiller dried grains (DDG), bran, poultry manure, mineral premix and molasses. This stage lasts 107 days with cattle weighing 514 kg. Per year 5,880 head are processed in highly technified slaughterhouses 138 km away from fattening enclosures.

On the other hand the fattening stage in the extensive chain takes place on pasture supplemented by a ration of 2 kg/head of 15% raw protein for the last 120 days. The stage lasts 213 days. Cattle at the end weigh 455 kg and are processed in low technology slaughterhouses 80 km away.

2.1.7. Transformation

Beef are slaughtered in modern and highly technological plants at 508 kg, a carcass yield of 59% is obtained. Organic residues, bone and tallow are turned into meal in a rendering plant and the manure is supplied to local farms. Residual water from the slaughterhouse is processed in a water-treatment plant. Carcasses are sectioned and deboned and vacuum-packed in polypropylene bags and placed in cardboard boxes. 12.1 tons of meat per year is transported to supermarkets distant on average 350 km.

In low technology slaughterhouses (extensive chain) livestock is processed at 450 kg. The carcass yield is 52%. Manure, blood, hooves and horns are thrown away into municipal dumps. Residual water is flushed into the local drainage system and bones and tallow are sold to a rendering plant. Fresh carcasses are taken to local butchers distant on average 7 km. 753,064 kg of carcasses are sold at the butchers' every year.

2.1.8. Marketing

Supermarkets receive a total of 278,351 kg of cut, packed meat. This is carved into beef cuts and sold to the final consumer. The packing consists of a polystyrene tray, a layer of PVC and a coated paper label. Supermarkets sell 24,818.3 kg of organic residue, which they sell to a rendering plant 15 km away. Meat is exhibited in refrigerated showcases. The power consumed in conservation, exhibition and cutting procedures was measured. Power used for lighting the sales-floor was not. Consumers cover 2.3 km (presumably in their own vehicle) to buy meat.

On the other hand, local butchers receive half carcasses, which they keep in cold storage and exhibit in refrigerated showcases. Loss through cut and deboning is 45%. Meat is sold in biodegradable poly-paper bags. 5,698 kg of organic residue is generated every year and sold to a rendering plant. The market is a local one and consumers walk to the butcher shops or take public transport.

2.1.9. Estimation of Greenhouse Gas Emissions

In the growth and development, pre-fattening and fattening stages on pasture no housing is necessary; therefore all manure is assumed to end up directly on the ground. The manure produced by livestock in the fattening process in pens is collected and spread on farmland belonging to pre-fattening farms, part of the intensive chain, 15 km away. The nitrogen excretion was estimated in order to calculate amounts of direct nitrous oxide, ammonia and nitric oxide emissions from the manure as well as indirect emissions of nitrous oxide through volatization and lixiviation, IPCC (2006).

2.1.10. Evaluation of Impact in LCA

Impact evaluation in LCA involved calculating the contributions of materials and fuels and the output at inventory phase in each of the environmental impact categories. The aforementioned categories are: climate change, acidification and human toxicity, all considered relevant in evaluating the environmental behaviors of obtaining animal products in a production chain. Impact evaluation was estimated using the CML 2000 method (Guinée et al., 2002), and SimaPro 7.3 LCA software from Pré Consultants. Greenhouse gas emissions, expressed in CO_2-eq units were quantified using the IPCC (2006) method and considering a temporal horizon of 100 years. Acidifying emissions are expressed in SO_2-eq and human toxicity in 1,4-DB-eq.

3. Results and Discussion

3.1. Evaluation of Life-Cycle Impact Results

In the intensive chain and in order of magnitude, the G & D herd used up more resources and produced more emissions in the categories of climate change impact and human toxicity. Pre-fattening demonstrated more acidification emissions in this chain. In extensive production the G & D stage is the greatest contributor to the impact categories. In both production chains enteric methane is the main contributor to climate change, followed by a substantial contribution from nitrous oxide obtained from manure biodegradation. In both chains, the fattening process is the one that least contributes to acidification and to climate change. This data coincides with reports from Pelletier et al. (2010), who compared three cattle production scenarios from G & D to finalization and observed the fattening process as the one, which impacts the least. In both chains, terms of acidification during the fattening stage showed the lowest levels. Pre-fattening shows the lowest impact for human toxicity in both production chains. In the intensive production chain, fattening is second to G & D for human toxicity emissions. In the extensive production chain G & D comes first and commercialization second in terms of human toxicity emissions (Figure 1).

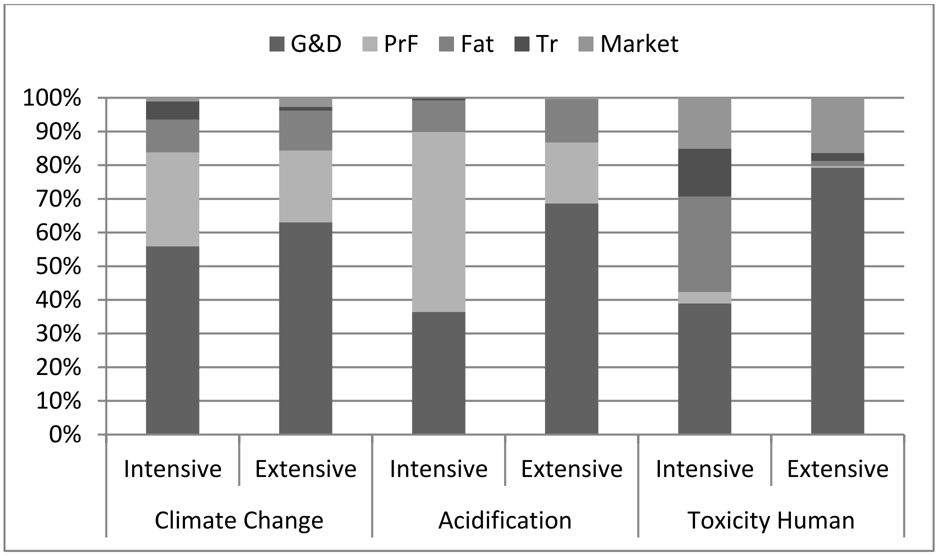

Figure 1. Total impact values shown in individual procedures by production chain.

3.2. Climate Change

Our results suggest that 1 kg of meat from Veracruz, Mexico produced in a chain based on intensive finishing systems and adopting modern technologies in the transformation and marketing stages has the potential to produce higher greenhouse gas emissions (17.80 kg CO_2-eq) than 1 kg of meat from an extensive beef production chain, including more traditional technologies in the transformation and marketing procedures. The main emissions are shown in Figure 2 where methane is the main contributor to climate change, with extensive productions emitting more methane (10.0 kg CO_2-eq) contributing to 63.3% of the total GHG when compared to the extensive chain (9.0 kg CO_2-eq) contributing to 50.7% of the total GHG. This result is consistent with another research showing that a higher quality diet for ruminants produces lower methane levels and increases the growth rate of livestock, therefore reducing methane emissions in the meat production life-cycle (Cederberg et al., 2009; Pelletier et al., 2010; Peters et al., 2010). Likewise, the higher levels of nitrous oxide in the intensive production chain are associated with manure management and production and processing of feed components in drylot-fed livestock.

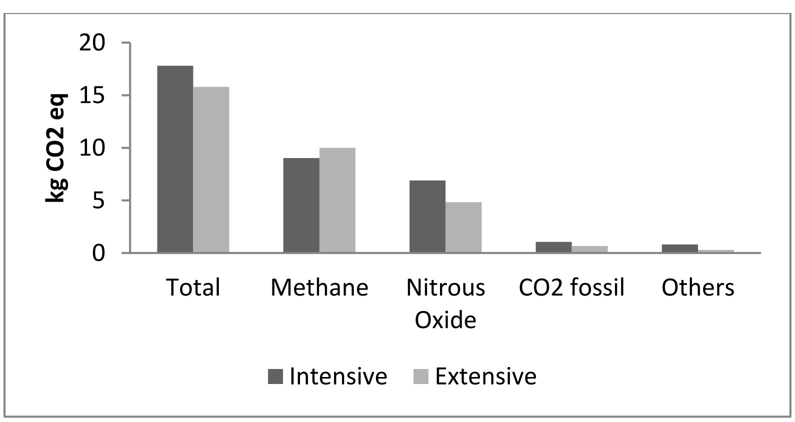

Figure 2. Comparison of greenhouse gas emissions by production chain.

3.3. Acidification

The potential for acidification in the intensive production chain was estimated to be greater than the extensive one (0.24 vs 0.13 kg SO_2-eq). This is directly related to the amount of ammonia produced when manure is accumulated in the confinement corral and then distributed over farmland; this result in greater ammonia volatilization as opposed to livestock on pasture (see Figure 3). The values obtained are consistent with those found by Roer et al. (2013) who used LCA to examine 1 kg of meat off the carcass from double purpose livestock. The higher emission of sulphur dioxide in the intensive production chain is a result of production and processing methods in the making of the feed during the feedlot stage. However, processing and marketing provide a smaller contribution compared to the production stages. This is in agreement with Meneses et al. (2012) who compared the milk production stage to the transport and packing.

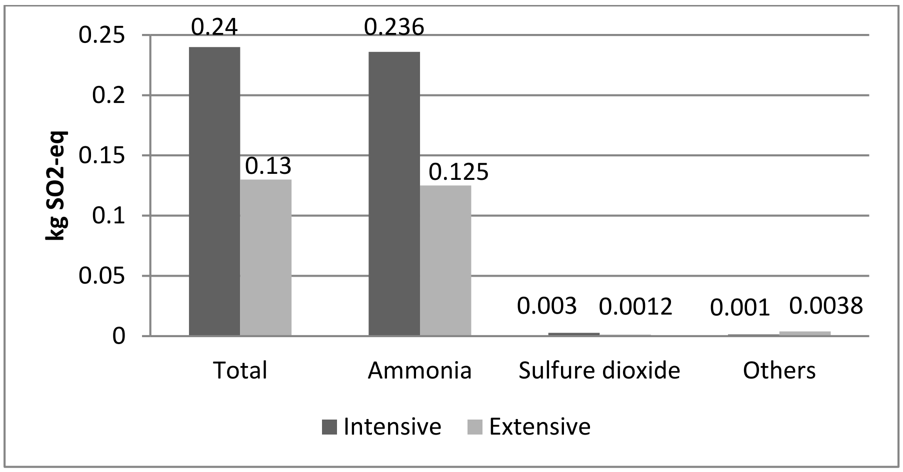

Figure 3. Contributing emissions to the acidification impact category by production chain.

3.4. Human Toxicity

Once again the intensive production chain shows higher potential for impact on human toxicity. In the case of the total potential emissions in both chains, polycyclic aromatic hydrocarbons show the highest potential emissions, 22.3% (0.13 kg 1,4-DBeq) in the intensive chain and 22.1% (0.06 kg 1,4-DBeq) in the extensive chain, as seen in Figure 4 due to the use of grain in drylot feed and food supplements during pre-fattening. This feed requires greater amounts of pesticides and fossil fuels for its production as well as more electricity during transformation and packing materials in the transformation and marketing stages. These results agree with Röös et al. (2013) who puts forth that an intensive beef production system has higher environmental impact potential due to greater quantities of agro-chemicals used.

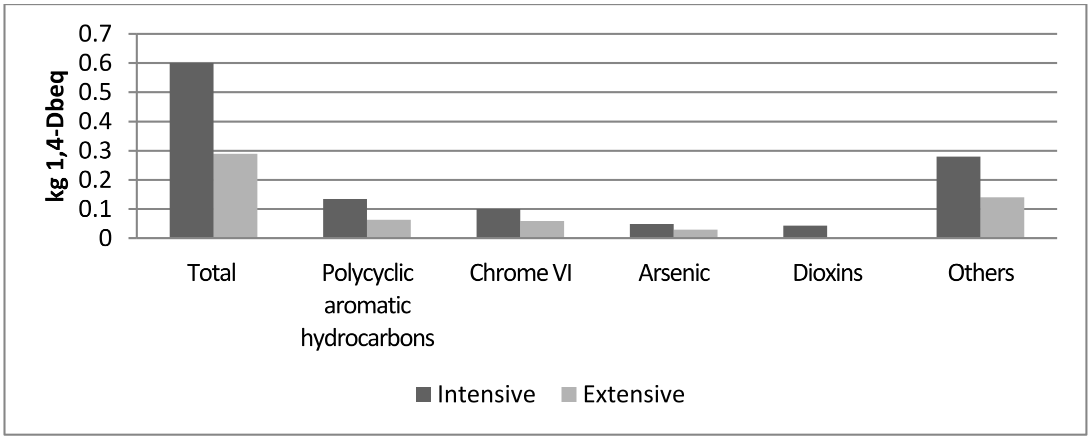

Figure 4. Contributing emissions to the human toxicity impact category by production chain

4. Conclusions

It was found that regardless of what systems of finishing, transformation and marketing are used, G & D is the main stage that contributes to the impact categories. This support the results of previous studies (Beauchemin et al., 2010; Pelletier et al., 2010) and is strongly related to the low fertility in bovine livestock compared to other species. In the present study cows produce one calf every 20 months and the maintenance of bulls and replacement calves must also be considered. However considering that calves come from double-purpose systems (milk production and veal production) the emissions generated were shared amongst both products, thus reducing the impacts from calf production which would be greater if the emissions were assigned solely to calf production; this coincides with other authors' findings (Roer et al., 2010).

The higher consumption of forage of livestock exclusively grown on pasture, for each kg of boneless meat produced, leads to higher methane emissions as a result of enteric fermentation; this is due to their lower feed conversion rate associated to the feeding system. However, it is important to consider the resources used to feed pasture-finished beef cattle are not competing with human nutrition, thus reducing the impact on the food and nutrition security due to cereals and legumes used in drylot finishing. Therefore, from an anthropocentric perspective, it is beneficial to obtain beef from pasture production chains.

The transformation stage carried out in slaughterhouses adopting modern technologies implies higher electricity requirements in order to process and store meat. In addition packing materials have an additional impact on beef production. However these systems allow for a much larger scale of production and transport of beef thus allowing satisfying the increasing demand for food, particularly in large urban settlements.

LCA has the potential to support decision making from a production chain perspective. This is extremely useful in the case of foodstuffs that involve relevant consequences on the environmental, economic and social necessities of the different stakeholders. Therefore further studies should integrate the social and economic dimensions of sustainability and the best way to effectively communicate the results to the consumer and other end users of researches on beef sustainability.

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The importance of regionalized LCIA in agricultural LCA – new software implementation and case study

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ABSTRACT

The necessity of performing regionalized LCA in agriculture derives mainly from the wide variety of existing farming systems and the influence of site-specific characteristics (e.g. climate, soil type, water availability) on the environmental impacts. Challenges on linking regionalized inventories and LCIA methods which use different spatial scales may be overcome by integrating GIS in the calculations. A new approach for GIS-based LCIA has been implemented in the open source software openLCA with the support of the USDA, where the calculation of regionalized characterization factors is parameterized using site-dependent and substance-dependent properties. The regional parameters are stored as attributes of each geolocalized element in shape files. The approach has been implemented in a case study on corn production in five US states. LCIA results from the Ecological Scarcity 2013 and enhanced Ecoindicator99 (EI99+) methods varied significantly between locations and, even more, depending on the spatial support used for the regional parameters.

Keywords: regionalized LCIA, openLCA, GIS, corn case study

1. Introduction

Environmental life cycle assessment (LCA) of agricultural systems commonly focuses on the impact categories land use, eutrophication, toxicity, acidification, climate change or depletion of abiotic resources (Bentrup *et al.* 2004; Payraudeau and Van der Werf 2005; Harris and Narayanaswamy 2009). The necessity of performing a regionalized life cycle impact assessment (LCIA) derives from the fact that the characterization factors in the majority of these impact categories are dependent on site-specific characteristics (e.g. population, soil types, climate factors, etc.). Appropriate spatial scales have to be selected for defining each of these different variables in order to minimize their spatial uncertainty (Manneh *et al.* 2010). However, this usually leads to a heterogeneity of spatial units within impact categories, which makes the calculation of the characterization factors at the resolution required by the inventory data more difficult (Gotway and Young 2002; Perveen and James 2009). Geographic differentiation of processes in an agricultural life cycle may also vary from global to farm level, usually prevailing site-specific modelling in data sets of the foreground system. Consequently, the ability to deal with multiscale systems without compromising the correctness of the results is fundamental in a regionalized impact assessment (Halog and Bortsie-Aryee 2013).

Geographic information systems (GIS) have been used by several authors to integrate spatial differentiation in LCA, not only for the calculation of regionalized characterization factors but also for creating site-specific inventories and matching each of these (Bengtsson *et al.* 1998; Geyer *et al.* 2010, Nuñez *et al.* 2010; Mutel *et al.* 2012; Liu *et al.* 2014). Moreover, Gerber *et al.* (2013) stated in a report of the food and agriculture organization of the United Nations (FAO) the convenience of using GIS for incorporating spatial heterogeneity into the modelling process and Halog and Bortsie-Aryee (2013) also recommended GIS as a powerful tool for supporting regionalized LCA.

Considering the importance of performing regionalized LCAs in activities such as agriculture and the feasibility of linking spatially differentiated inventories and impact methods using GIS, the integration of the functionality to conduct fully regionalized LCIA should be a priority for LCA tool developers. This paper aims to present a new implementation approach for GIS-based regionalized LCA developed in the open source software openLCA. The application of this new feature in a case study on corn production in five different states of the United States is also examined.

2. Methods

2.1. Extension of geographic information in process data sets

In contrast to the tendency in impact assessment methodologies where more detailed regions (e.g. watersheds, ecozones, etc.) are used for the definition of the characterization factors, most commonly used LCA databases (e.g. ecoinvent 3, GaBi 2013, ELCD3) continue using country or group of countries locations only. However, spatial differentiation of agricultural systems usually requires higher spatial resolution geographies due to the considerable diversity of farming systems (Pradeleix *et al.* 2012). Therefore, the feature for specifying the geographic information of the process in openLCA needed to be enhanced to allow the LCA practitioner to differentiate between site-specific locations and to integrate GIS information in the data sets.

Data exchange formats most commonly used in LCA tools (i.e. EcoSpold1, ILCD, EcoSpold2) are all based on extended markup language (XML). The geographical information included in them has evolved from the limited location code of EcoSpold 1, to the addition of a latitude and longitude pair in ILCD and finally to the integration of keyhole markup language (KML) geographic descriptions in EcoSpold2 (Mutel 2009). As the geospatial information stored in KML files can be used by GIS, a KML editor for the processes was added in openLCA (Fig. 1). In this editor, the process location can be defined using three geometrical primitives (i.e. point, line and polygon). The resulting geographical information, characterized by the object's geometry and a set of coordinates, is known as the spatial support of the dataset (Plumejeaud *et al.* 2010). This spatial support can be easily imported and exported using a slightly modified ecospold 2 format, where KML data is also linked to the process id.

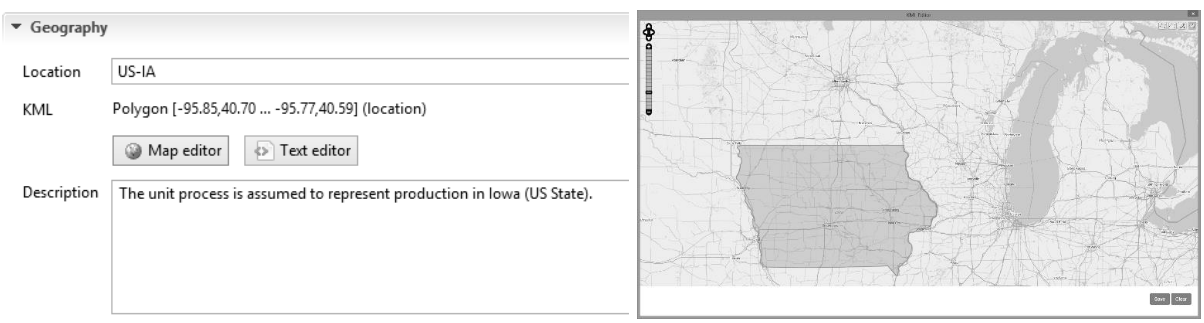
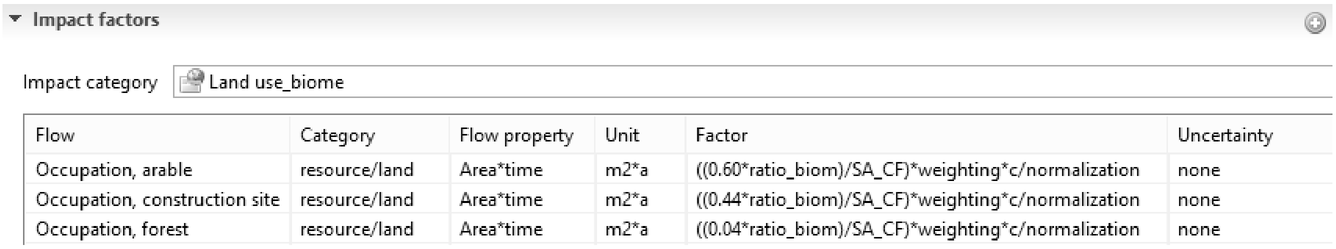


Figure 1. Extended geographic information in a process data set (left) and KML editor (right) (openLCA 1.4)

2.2. Parameterization of regionalized LCIA methods

The availability of impact assessment methods providing characterization factors not only on a site-generic scale but also on a country and sub-country level or even for very high spatial resolution geographies has increased considerably in recent years (e.g. EDIP 2003, EcoIndicator 99+, ImpactWorld+, etc.). The increase in the amount of data resulting from this spatial differentiation of the factors has been overcome by using GIS files as databases of the site-specific characterization factors. However, concerns about integrating all this GIS data in the LCA software without compromising the computing power of the tool fostered the development of a new concept for dealing with regionalized LCIA in openLCA. In this new approach, the mathematical functions used in the LCIA models are parameterized in order to differentiate between site-generic (i.e. dependent on substance properties) and site-specific (e.g. population density) variables. The resulted functions per substance and impact category are defined in the openLCA LCIA method editor (Fig. 2), whereas the data for the regional characteristics are stored in shape files (i.e. GIS vector data) as attributes of each geolocalized element. The data from the different attributes extracted during the import of the shape files can be bound to the parameters used in the characterization factors functions and a default value to be used with site-generic inventory data can also be defined (Fig. 3). Moreover, uncertainty of each characterization factor and parameter can be included. As the shape files are stored in the LCA database, they can be easily exported with the database.

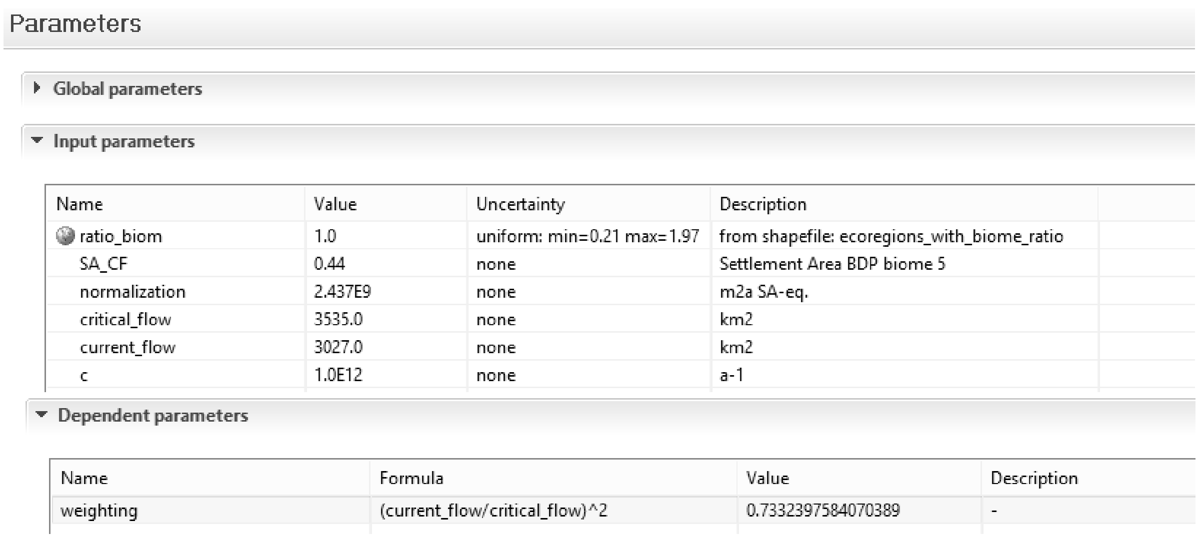


Impact factors

Impact category: Land use_biome

Flow	Category	Flow property	Unit	Factor	Uncertainty
Occupation, arable	resource/land	Area*time	m2*a	((0.60*ratio_biom)/SA_CF)*weighting*c/normalization	none
Occupation, construction site	resource/land	Area*time	m2*a	((0.44*ratio_biom)/SA_CF)*weighting*c/normalization	none
Occupation, forest	resource/land	Area*time	m2*a	((0.04*ratio_biom)/SA_CF)*weighting*c/normalization	none

Figure 2. Impact factors tab in the new LCIA method editor (openLCA 1.4)



Parameters

Global parameters

Input parameters

Name	Value	Uncertainty	Description
ratio_biom	1.0	uniform: min=0.21 max=1.97	from shapefile: ecoregions_with_biome_ratio
SA_CF	0.44	none	Settlement Area BDP biome 5
normalization	2.437E9	none	m2a SA-eq.
critical_flow	3535.0	none	km2
current_flow	3027.0	none	km2
c	1.0E12	none	a-1

Dependent parameters

Name	Formula	Value	Description
weighting	(current_flow/critical_flow)^2	0.7332397584070389	-

Figure 3. Parameters tab in the new LCIA method editor (openLCA 1.4)

2.3. Regionalized LCIA calculation framework

The matrix-based calculation in openLCA needed to be extended to include the geographic variable in the inventory, impact assessment and results matrices. Moreover, in order to handle geospatial data, the open source Java code library GeoTools (2013) was integrated in the software.

The first step in the regionalized LCIA calculation in openLCA is to obtain the inventory matrix *G* which contains all the emissions and resources consumed per process and, consequently, per location as defined in each data set. Then, the regionalized LCIA matrix *C* containing the characterization factors per elementary flow and location in the inventory and per impact category in the method is calculated. To this end, the area of each spatial unit where the regional parameters are characterized intersected by each geographic feature of the inventory is determined. Then, a weighted average value for each parameter is obtained applying equations 1-3, depending on whether a point, line or polygon is used for defining the process location (Fig 4). Once the parameters values are calculated, the formulas in the method are evaluated resulting in the matrix *C*. Finally, the multiplication of matrices *C* and *G* provides the matrix *R* of regionalized impact assessment results.

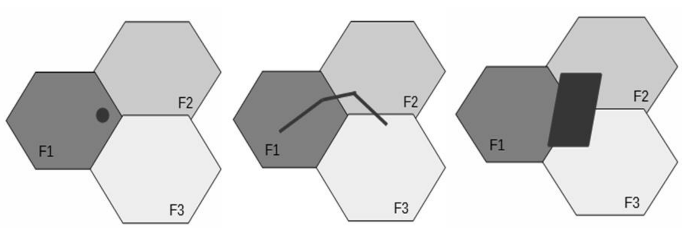


Figure 4. Schematic representation of the calculation of the intersected spatial units from the LCIA method (F_i) by the process geometries point (left), line (center) and polygon (right).

$$p_{Fi} = p \qquad \text{Eq. 1}$$

$$\frac{\sum_{i=1}^{n}\left(p_{Fi}L_{Fi}\right)}{\sum_{i=1}^{n}L_{Fi}} = p \qquad \text{Eq. 2}$$

$$\frac{\sum_{i=1}^{n}\left(p_{Fi}A_{Fi}\right)}{\sum_{i=1}^{n}A_{Fi}} = p \qquad \text{Eq. 3}$$

where L_{Fi} and A_{Fi} represent the length of the line and the area of the polygon contained in each feature Fi of the shape file (i.e. spatial unit where the regional parameters are characterized), respectively.

2.4. Case study inventory and preliminary impact assessment data

Life cycle models of corn production using cradle-to-farm gate as system boundary were created for the US states of Illinois, Iowa, Nebraska, North Dakota and Minnesota. The LCA Digital Commons (2013) database developed by the United States Department of Agriculture (USDA) and National Agricultural Library (NAL) was used for the activities occurring within the farm. The rest of the life cycle was completed with unit processes from ecoinvent v.2.2 and GaBi 2012 database extension XVII - full US. The KML information of the five states was obtained from the US Census Bureau (2014). The functional unit of the different product systems is the production of 1kg of "corn grain, at harvest in 2005; at farm; 85%-91% moisture".

The method Ecological Scarcity 2013 (Frischknecht and Büsser Knöpfel 2013) was used for calculating the environmental impacts of the product systems, focusing mainly on the categories land use and freshwater consumption as they include regionalized characterization factors at different spatial resolution scales. The regional eco-factors are obtained applying to the characterization factor (*K*) a weighting factor calculated on the basis of the current (*F*) and critical annual flows (F_k) from a specific region and normalized to the annual flow in Switzerland (F_n^{CH}) (Eq. 4). In the case of freshwater consumption impact category, the weighting factor can be calculated by the ratio of water withdrawal to renewable water supply (i.e. the water scarcity), considering that the critical flow is 20% of the water supply (Eq. 5). The method developers provide a list of the resulting water scarcities on global and country levels, as well as a Google™ earth layer with values per watershed (Treeze Ltd 2014). This GIS data was adjusted to the shape file format required by openLCA (i.e. including the water scarcity values as attributes of the different features) using the free, open source software QuantumGIS. In addition, a shape file containing the country specific eco-factors and water scarcity ratios was created.

$$Eco-factor^{\mathrm{Region\_1}} = K \cdot \frac{1 \cdot UBP}{F_n^{CH}} \cdot \left(\frac{F^{\mathrm{Region\_1}}}{F_k^{\mathrm{Region\_1}}}\right)^2 \cdot c \qquad \text{Eq. 4}$$

$$Eco-factor^{\mathrm{Region\_1}} = K \cdot \frac{1 \cdot UBP}{F_n^{CH}} \cdot \left(\frac{water\_withdrawal^{\mathrm{Region\_1}}}{water\_supply_{renewable}^{\mathrm{Region\_1}} \cdot 20\%}\right)^2 \cdot c \qquad \text{Eq. 5}$$

Where *c* is a constant (i.e. 10^{12}/a) and *UBP* is eco-point, the metric used for expressing the environmental impact assessed.

Regarding land use impact, the Biodiversity Damage Potentials (BDPs) of biome 5 (i.e. temperate coniferous forests) from de Baan *et al.* (2012) are weighted with the ratio of species densities from Kier *et al.* (2005) to biome 5 in order to obtain biome-specific BDPs. "Settlement area" (SA) land use is selected as the reference "substance" for the determination of the characterization factors (*K*) that derive of the resulting BDPs (Eq. 6). The regional eco-factors are then calculated applying equation 7. This equation is defined per relevant elementary flow in the openLCA LCIA method editor as shown in figures 2-3. For a non-regionalized LCIA, the eco-factors of biome 5 are used (i.e. default ratio of species densities equal to 1). A shape file containing the features of the 14 biomes considered in the method with their correspondent ratio to the biome 5 as attribute was created for openLCA.

$$K^{biome\_i} = \frac{BDP^{biome\_i}}{BDP\_settlement\_area\_biome5} = \frac{BDP^{biome5} \cdot ratio^{biome\_i}\_to\_biome5}{BDP\_settlement\_area\_biome5} \qquad \text{Eq. 6}$$

$$Eco-factor^{\mathrm{Re}gion\_1} = K^{\mathrm{Re}gion\_1} \cdot \frac{1 \cdot UBP}{F_n^{CH}} \cdot \left(\frac{F}{F_k}\right)^2 \cdot c \qquad \text{Eq. 7}$$

Moreover, the impact due to water consumption was also measured using the enhanced EcoIndicator99 (EI99+) method (ETZ Zürich 2014), which provides characterization factors at the watershed and country levels for the midpoint indicator WSI (i.e. water stress index), as explained by Pfister *et al.* (2009).

3. Results

3.1. GIS-based regionalized LCIA implementation in openLCA

The implementation of the new features and calculation framework for regionalized LCIA in openLCA was performed successfully (Fig. 1-3) and without affecting significantly the calculation time required when a single line, point or polygon was used in the geographical specification of the process. However, the ability of dealing with multi-polygons (e.g. countries containing continental and overseas territories) must be refined. Moreover, additional enhancements such as the integration of regionalized LCIA in the project level (i.e. comparison of different product systems) will also be made. The already implemented features were included as experimental in the first release of openLCA 1.4 in June 2014.

3.2. Preliminary case study results

The characterization factors calculated by openLCA varied largely between the states assessed and, even more, within the same location depending on the spatial units used for the regional parameters. Table 1 presents the results for the three impact categories previously described in section 2.4. The WSI from EI99+ was for three states one order of magnitude lower than the US mean value, while Nebraska had a slightly higher value than the country average. The eco-factors for arable land use calculated using biome scale varied also 25.9% between North Dakota, the state with the lowest value, and Minnesota, the one with the highest eco-factor. As the states assessed in this study were located within biomes 4 (i.e. Temperate broadleaf and mixed forests) and 8 (i.e. Temperate grasslands, savannas and shrub lands), it is expected that characterization factors for other states with most sensitive ecoregions (e.g. California) will present even higher differences respect to the generic value (i.e. biome 5).

Table 1. Characterization factors for the EI99+ midpoint category WSI and the Ecological Scarcity 2013 categories land use (i.e. arable land) and freshwater consumption calculated with openLCA 1.4 (beta 6) for five states of the USA, using different spatial resolutions (i.e. country, watershed, biome)

	WSI (m3/m3) – EI99+		Eco-factor for land use, arable (UBP/m2a SA-eq.)		Eco-factor for freshwater consumption (UBP/m^3)		
Location	**Country**	**Watershed**	**Generic**	**Biome**	**Global**	**Country**	**Watershed**
Illinois	4.99E-1	2.81E-2	420	360	610	232	109
Iowa	4.99E-1	5.31E-2	420	325	610	232	7376
Minnesota	4.99E-1	2.89E-2	420	397	610	231	4717
Nebraska	4.99E-1	5.94E-1	420	294	610	232	62532
North Dakota	4.99E-1	1.11E-1	420	294	610	231	45828

The impact category freshwater consumption from the method ecological scarcity 2013 deserves a deeper analysis as huge differences exist between the results obtained using the country-level and the watershed spatial resolution. As reported by Frischknecht *et al.* (2009), the country water scarcity values are obtained from the AQUASTAT database of FAO[1], whereas the data for a deeper regionalized assessment is retrieved from the database of the University of New Hampshire[2], whose indicators are used in the World Water Assessment Program of UNESCO. Specifically, the indicator I4 (i.e. mean annual relative water stress index) with values per grid cell was used to generate unweighted aggregated values per watershed (Fig. 5). The resulting averages per water basin led to extreme water scarcities (i.e. >1) in the areas intersected by some states used in the case study. Consequently, the eco-factors calculated by openLCA with data at the watershed level are much higher than those obtained with the country data.

Figure 5. Mean annual relative water stress index per watershed adapted from Treeze Ltd (2014) for its use with the ecological scarcity method 2013

The results for impact categories for which no regionalized characterization factors were available varied also between the product systems analyzed, as different inventory results were obtained due to the state-specific data sets used for the farm activities. Figure 6 presents the LCIA results of the Ecological Scarcity 2013 method. It can be observed that Nebraska is the state with highest environmental impacts for most of the impact categories, except for land use where North Dakota and Minnesota had the higher impacts. This reinforces the idea that the availability of regionalized inventories when performing site-dependent LCAs is as important as having regionalized characterization factors. For instance, if the same inventory had been used for all states, Minnesota and not North Dakota would have been the farming system with highest impacts due to land use.

[1] http://www.fao.org/nr/water/aquastat/main/index.stm

[2] http://wwdrii.sr.unh.edu/download.html

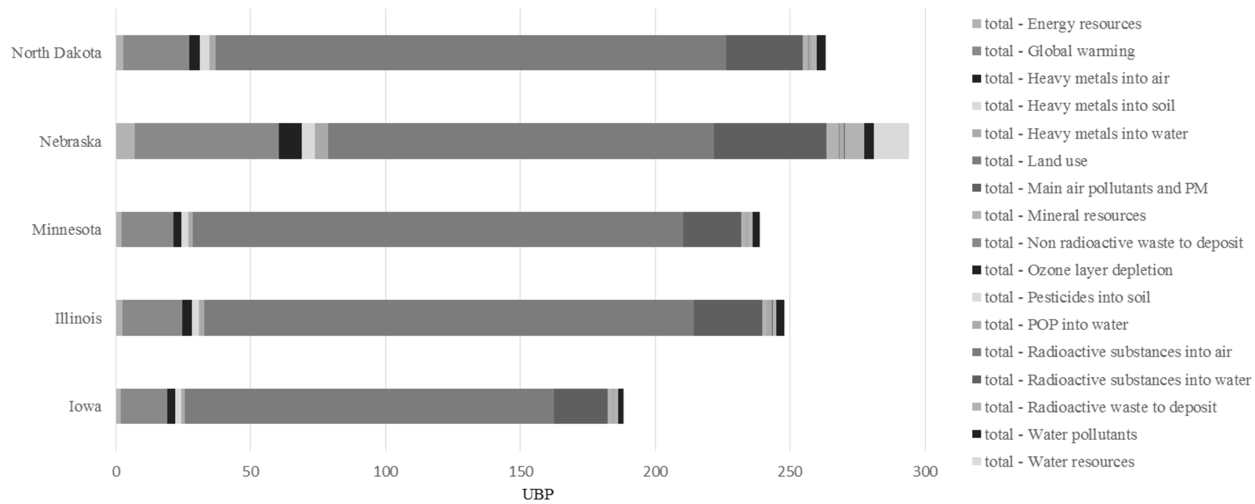


Figure 6. 1kg corn grain production LCIA results in UBP for the Ecological Scarcity 2013 method (regionalized assessment of land use and water resources impact categories). Results calculated for farming systems in Illinois, Iowa, Minnesota, Nebraska and North Dakota.

4. Discussion

The variety in the results obtained from the calculation with different spatial scales showed the convenience of using non-aggregated data for the characterization of the regional parameters in the LCIA methods. However, for some factors, division at the cell level (e.g. per square meter) might lead to missing some important regional conditions relevant to the effect of the emission or resource consumption. For instance, Frischknecht *et al.*, (2009) described that water stress modeling at the cell division level does not consider hydrological conditions; the result of this is that areas suffering from great water stress lack water stress values in the model because no water extraction takes place there, (i.e. large parts of the Sahara). Thus, it is necessary to determine the most suitable spatial resolution to use for each parameter so that all the relevant regional properties are considered and their spatial uncertainty is minimized. The use of weighted aggregations might be useful for avoiding misleading values such as in the case of the WSI of Figure 5. To this end, available data on background emissions or likely geographical distribution of emissions (i.e. emission proxies) could be used to determine the areas where the probability of occurring the impact is higher (ImpactWorld+ 2014). Likewise, geographical distributions of the processes might be also applied when determining the location of each activity (Mutel *et al.* 2012). The uncertainty derived from these spatial distributions might affect not only the inventory results but also the values of the regionalized characterization factors calculated in the software. Therefore, the addition of the process spatial uncertainty to the uncertainty distribution of the dataset exchanges and of the LCIA characterization factors should be considered.

It should also be borne in mind that in the current approach the data from the regional parameters used for calculating the weighted average is only that contained in the intersected area by the process geometry. However, as the full impact from a source can cover areas extending several hundred to thousand kilometers (Potting and Hauschild 2006), using site-specific locations in the inventory might reduce the accuracy of the results. Therefore, it might be necessary to analyze the feasibility of enhancing the current implementation approach to also include transport pathways of emissions when calculating the regionalized characterization factors.

Moreover, the seasonal variation of the regional parameters is currently dismissed as annual averages are commonly used, for example, in the water scarcity ratios calculation. However, this might affect the impact results considerably as some crops are only grown during specific times of the year. Therefore, future advances in the software implementation of regionalized LCIA should also integrate the temporal variable.

Finally, LCIA results obtained from spatially heterogenic systems, i.e. multi-scale systems, should be carefully interpreted: processes using generic characterization factors, which tend to be higher than site-dependent factors (Table 1), might have higher contributions to the overall impact than expected. Therefore, the decision of performing a regionalized LCIA must be along with the goal of the study whilst bearing in mind the added complexity to the analysis of the results derived from this type of assessment.

5. Conclusion

Considering the high variability of LCI and LCIA results between the different locations analyzed, regionalized assessments for finer spatial resolutions than countries should be conducted both in the inventory and impact assessment phases for agricultural systems. To this end, LCIA method developers should provide the regional parameters used for calculating the characterization factors using suitable spatial resolution. Higher transparency in the calculations applied for the different impact categories will also help to implement their methods in openLCA. Likewise, LCA database providers should consider the necessity of modeling the data sets at higher spatial resolutions than national or supranational scales. The currently integrated approach in openLCA allows to perform fully regionalized LCAs without increasing considerably the complexity of the calculations. However, this first step in the implementation must be followed by further enhancements which will allow to minimize the uncertainty and increase the validity of the impact assessment results.

6. Acknowledgments

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Carbon footprint along the Ecuadorian banana supply chain: Methodological improvements and calculation tool

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ABSTRACT
Bananas are among the most important food and agricultural commodities worldwide, being Ecuador its main exporter as it represents over a third of the global banana exportation. A detailed carbon footprint assessment of the complete Ecuadorian banana value chain has been carried out. Special attention was paid to the adaptation of the emission factors to the local conditions, especially in those activities that were expected to contribute most to the whole footprint. The carbon footprint of Ecuadorian bananas from cradle-to-grave was 1.25 ton CO_2e per ton of banana at the consumption point in Spain. Farm stage was identified as the main contributor (22.1 %) and significant differences were found between the environmental performance of conventional and organic farms, mainly due to the use of synthetic fertilizers in the former and the related N_2O emissions. An Excel calculation tool was developed, to allow Ecuadorian stakeholders to evaluate different operating conditions.

Keywords: fruit, food, cradle-to-grave, value chain

1. Introduction

Food production and consumption has been proven as an important contributor to environmental degradation, being responsible from 20 to 30% of the impacts of private consumption (Tukker et al. 2006).

In developed countries, consumers start demanding food produced with minimal environmental losses (de Boer 2003) and increasingly base their purchase decisions on environmental indicators shown in food packaging. Within these indicators, the carbon footprint (CF) of a certain product, defined as the total greenhouse gas (GHG) emissions over its whole life cycle, expressed as CO_2 equivalents (Carbon Trust 2012), has achieved widespread development over the past decades, because of being very intuitive and easily understandable for non-expert users, which facilitates the diffusion of the results (Weidema et al. 2008).

Provided that agriculture accounts for about 14% of global GHG emissions (IPCC 2007), due to the recent technological developments that farms have experienced, vegetal food production has become an important contributor to climate change. Thus, a substantial number of carbon footprint assessments of plant-based products have been published recently (Espinoza-Orias et al. 2011; Gan et al. 2012; Röös et al. 2010). Within these studies of plant products, the CF assessment of tropical fruits consumed in Europe is of great interest, since the emissions related to transoceanic transportation from production countries to consumption sites must be added to those associated with the farming stage (Brito de Figueirêdo et al. 2013; Ingwersen 2012; Sim et al. 2007).

This study analyzes the carbon footprint of the whole value chain of bananas grown in Ecuador, the world's largest exporter (A.E.B.E. 2011), and consumed in Spain. To the best of our knowledge there are five studies that have already evaluated the carbon footprint of banana production (Eitner et al. 2012; Iriarte et al. 2014; Lescot 2012; Luske 2010; Svanes and Aronsson 2013). All these papers place their final destinations in northern Europe, and only one includes consumption stage (Svanes and Aronsson 2013). In particular, Ecuadorian plantations were studied by Eitner et al. (2012) and Iriarte et al. (2014). In the first one, CF calculations were conducted using the Footprint Expert Tool[1], in which the assessment of farm emissions is based on average European data and process calculation is not as transparent as desirable as emission factors are not published, while the second one only considers data collected in a single farm in El Oro region.

This study starts from the identification of the main contributions reported and the weaknesses found in existing literature, to perform a more adapted assessment to the actual conditions considered here. Firstly, data from farming stage have been collected from a set of plantations located in the three provinces with higher banana yields (Guayas, Los Ríos and El Oro (A.E.B.E. 2011)); secondly, the assessment of nitrous oxide (N_2O) emis-

[1] http://www.carbontrust.com/software#footprintexpert

sions from fertilized soils has been adapted to tropical climate, while all the available studies on banana production used default values for temperate climate; and finally, a detailed revision of the emission factor for overseas transport has been carried out in order to better represent the real transportation mode of the Ecuadorian bananas.

2. Methods

To determine the carbon footprint of the whole value chain of Ecuadorian bananas, the methodologies defined by ISO 14067 (ISO 2013) and PAS 2050:2011 (BSI 2011b) have been followed.

2.1. Functional unit and system boundaries

A cradle-to-grave assessment has been performed, including all the stages shown in Figure 1, from banana farming in Ecuador to consumption phase in Spain, which complements the available studies with final destinations in countries of northern Europe, such as Germany or Norway.

A ton of banana arriving to the consumption stage (S8) has been chosen as the functional unit (FU). One ton of banana is the FU normally selected by other authors but not always located at the end of the value chain, i.e. at the consumers' hands, so results are not directly comparable due to food losses along each of the chain stages.

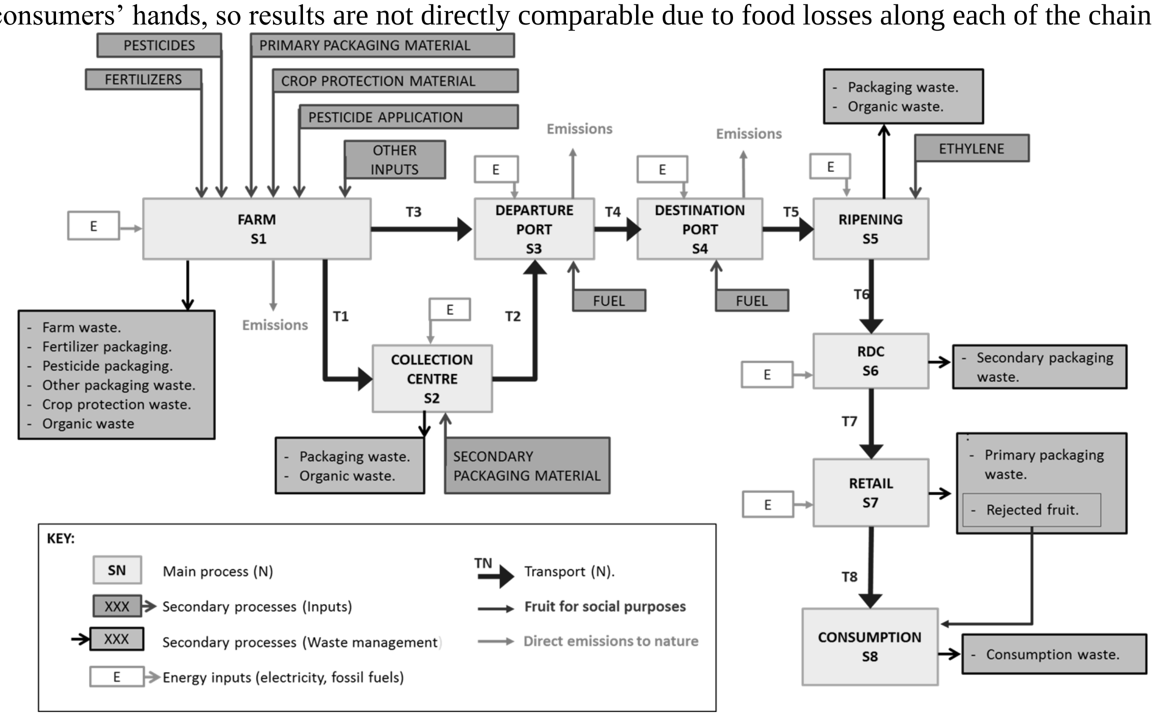

Figure 1. The banana supply chain under assessment.

An exhaustive data collection process has been carried out for obtaining inventories of inputs and outputs of each of the stages under study. The transport associated to the provision of secondary processes (inputs/outputs) has been excluded, with the exceptions of those transports that were expected to have a meaningful contribution to the whole carbon footprint: the provision of fertilizers and pesticides in S1, as the great majority come from Europe, and the waste transport of municipal solid waste generated in S8, due to their relatively high weight.

The GHG emissions arising from the production of capital goods have been included in the assessment when available, i.e. the manufacture of the vehicles used in transportation systems. The emissions related to the production of those capital goods having a very long lifetime (i.e. buildings, cableway, irrigation system…) have not been considered, provided that the corresponding share of those emissions to the functional unit is negligible.

As recommended in the aforementioned guidelines (BSI 2011a), neither biogenic carbon, and the delayed emissions associated to its storage in food products, nor emissions from land use change have been taken into account, as no change in the land use of the plantations has took place in the last 20 years.

For the determination of nitrous oxide emissions from fertilized soils, the IPCC guidelines (IPCC 2006) were used, having been adapted to the characteristics of the Ecuadorian region, as shown later in this document.

In order to keep uniformity and to make the results comparable, the Ecoinvent database (Frischknecht et al. 2007) has been used in the majority of the cases for the provision of the background emissions factors, being modified when necessary to adapt them to the Ecuadorian conditions, and complemented if required with some other sources as detailed in the next section.

2.2. Data sources

To characterize the farm system (S1), primary data was obtained from seventeen Ecuadorian plantations, of which nine were organic farms and the remaining eight, conventional farms. This classification was meant to allow establishing differences in the carbon footprint among both farm production systems provided that, for many foods, the environmental impacts of organic agriculture are lower than for the equivalent conventionally-grown food (Foster et al. 2006).

A questionnaire has been prepared and interviews with those responsible for the plantations have been performed. The data collected were, among others: general data of their planting site (coordinates, area, annual yield), use of crop protecting materials (type and amount of each item), use of fertilizers and pesticides (type and amount of each product), electricity and fuel consumption (pumps, irrigation systems, aerial and terrestrial spraying of pesticides), use of packaging materials (cardboard boxes and plastic bags for banana packing) and waste generation (amount and destination of the residues).

Given the impossibility of summarizing the vast amount of data collected by means of these questionnaires, Table 1 shows, as an example, the main inputs of the S1 system for some similarly sized farms.

Table 1. Summary of the main inventory data gathered for S1, for medium-sized farms.

Inputs	**Organic**				**Conventional**		
Crop protection material							
Cover (u)	49.62	60.05	39.06	55.60	42.63	70.10	49.10
Bow tie (u)	-	-	-	-	85.26	140.20	98.20
Banana hand cover (u)	-	-	-	-	-	-	39.28
Tape (u)	-	60.05	39.06	55.60	42.63	70.10	-
Nun's neck (u)	99.23	102.94	55.80	148.27	170.53	198.62	-
Packaging material							
Cardboard box (u)	55.13	55.13	55.13	55.13	55.13	55.13	55.13
Plastic bag (u)	55.13	55.13	55.13	55.13	55.13	55.13	55.13
Rubber (u)	55.13	55.13	55.13	55.13	55.13	55.13	55.13
Label (u)	1985	1874	1874	1985	1874	1874	1323
Glue (kg)	0.36	0.22	0.22	0.28	0.22	0.28	0.24
Pesticides							
Fungicide (l)	0.98	0.36	0.21	0.47	0.01	0.08	0.13
Herbicide (l)	-	-	-	-	-	-	0.09
Growth regulator (l)	0.28	0.24	0.01	0.10	-	-	-
Insecticide (l)	-	-	-	0.10	-	-	-
Unspecified pesticide (l)	0.07	0.04	0.21	0.28	-	-	-
Other compounds (kg)	-	-	-	-	0.19	0.35	0.33
Fertilizers							
Average fertilizer from algae (kg)	-	-	-	0.18	-	-	0.10
Lime (from carbonation) (kg)	23.85	11.93	-	-	3.85	1.52	-
Potassium chloride (kg)	-	-	-	-	11.54	17.32	2.51
Compost (kg)	67.27	35.78	17.55	-	-	-	-
Ammonium nitrate (kg)	-	-	-	-	3.85	4.23	10.02
Diammonium phosphate (kg)	-	-	-	-	-	3.23	2.51
Ammonium sulfate (kg)	-	-	-	-	3.85	14.04	2.51
Calcium sulfate (kg)	-	-	-	-	3.85	-	-
Potassium sulfate (mineral) (kg)	50.09	19.22	23.40	65.74	-	2.92	-
Poultry manure (kg)	-	-	-	59.16	-	-	-
Magnesium sulfate (kg)	-	-	-	-	-	0.45	-
Zinc sulfate (kg)	-	-	-	-	-	3.38	-
Urea (kg)	-	-	-	-	7.69	15.43	4.38
Other fertilizers (kg)	-	-	-	-	-	1.69	11.90
Energy							
Electricity (kWh)	-	7.61	-	1.80	1.16	11.81	7.68
Diesel (l)	21.19	19.38	46.68	12.76	25.53	30.52	3.48
Petrol (l)	3.82	1.93	4.02	1.99	1.73	1.73	10.95

A brief questionnaire has also been defined for cardboard boxes and plastic protection elements manufacturers, as those products are produced within the country, and information from two companies has been gathered in order to adapt the Ecoinvent datasets to the Ecuadorian productive systems.

Depending on their farm of origin, banana boxes can be shipped directly to the expenditure port (T3) or through a collection center (T1+T2). Data regarding T1, T2 and T3 transports, such as the type, size and load percentage of the trucks that transport bananas from the farm to the departure port in Ecuador, have also been gathered from farmers. When fruit was sent directly from the farms to the port (T3), refrigerated container trucks were used, which were loaded at the farms and kept closed until their arrival to Europe, while when fruit was sent to a collection center before its transport to the departure port, conventional trucks of different sizes were used in T1, and the type of truck (conventional or refrigerated) depended on the transport distance in T2.

To establish the energy requirements at the collection center (S2), an average facility has been considered (Martínez 2011), and a 0.20% of losses at this stage has been assumed.

There are two main harbors in Ecuador from which bananas are sent overseas: Puerto Bolivar and Guayaquil harbors (S3). Rotterdam (The Netherlands) has been selected as destination port (S4), as it is the main European destination of the shipping companies interviewed. Energy-related GHG emissions from ports have been determined based on data of a Chilean Port (Terminal de Puerto Arica 2011) for S3, and of several European ports (GreenCranes 2013) for S4.

Regarding overseas transport (T4), a weighted average has been calculated to establish the distance to be covered by the fruit (Searates 2013), based on the percentage of the total export of banana represented by each of the two Ecuadorian ports (A.E.B.E. 2011). The characteristics of the vessels used (type, size, load percentage in both trips (outbound and return) were provided by two shipping companies (Maersk Line and Sea Trade) responsible for the transoceanic transportation of this fruit.

The stages of ripening, distribution center, retail and consumption have been assumed to take place in Spain. Thus, once the fruit has arrived to Rotterdam, it is transported by refrigerated trucks (T5) to a ripening facility in Madrid (S5). Distances covered by all terrestrial transports were determined using Google Maps (2013).

At S5, fruit is ripened by controlled temperature and the addition of ethylene. Several bibliographic sources were used to determine electricity use (Eitner et al. 2012; Luske 2010; Svanes and Aronsson 2013), ethylene consumption (Svanes and Aronsson 2013), and the amount of fruit rejected at quality control (Luske 2010).

After ripening, cold chain ends and fruit is transported to Regional Distribution Centers (RDCs) by non-refrigerated trucks (T6). At RDCs (S6), secondary packaging is removed, and bananas are stored at room temperature until they are sent to retailers (T7). To model T6 and T7 transports, some of the more important RDCs for food products in Spain, and the main cities located in their areas of influence, were selected in order to guarantee the coverage of the whole country.

At retail stores (S7), primary packaging materials are removed (i.e. plastic bag and cardboard box) and fruits are sold at room temperature. Inventory data of the retail system was determined based on the distribution of fruit sales in Spain into different store categories (ICE 2011), and the energy requirements of each of them (Tassou et al. 2010). Regarding fruit waste at this stage, it has been estimated that only 2.24% of the bananas that reach this stage are sent to waste management, based on Spanish statistics (MAGRAMA 2012).

Once the fruit leaves the store, it is transported to households (T8) either on foot (90%) or by car (10%). The ending stage of the value chain is fruit consumption (S8). In Europe, bananas are stored at room temperature and eaten raw, being the main residue here the banana peel (40% of the total weight). Besides, the input and output flows associated to the ingestion and posterior excreta of the fruit have also been included (Muñoz et al. 2008). It has been assumed that all bananas reaching this stage are consumed.

It should be noted that, for all waste generated in the systems located in Spain (S5, S6, S7, S8), it has been assumed that waste management follows the current distribution in the country for each type of residue, based on Spanish statistics (INE 2011).

2.3. Adjustment of emission factors

To perform the carbon footprint assessment of the banana value chain, some methodological adjustments have been made to adapt the emission factors to the actual conditions of the study, which are described below.

2.3.1. N_2O emissions from managed soils

According to the IPCC guidelines (IPCC 2006), direct and indirect N_2O emissions arising from N fertilized soils can be determined from the amount of anthropogenic N inputs (i.e. synthetic or organic fertilizers, manure application, agricultural waste) through the use of emission factors.

On the one hand, to determine the N_2O direct emissions, a default value of 0.01 for EF1 is suggested, meaning that 1% of the applied nitrogen is emitted to the atmosphere as N_2O-N. Nevertheless, this factor is adapted to temperate countries, and independent of soil texture. In this study, specific emission factors for tropical climates have been used (Table 2), adapted to the three main types of existing soils in the area (MAGAP 2002).

Table 2. Emission factors for tropical countries used in determining N_2O direct emissions.

Soil texture	EF1 (%)	Source
Loamy (medium)	2.91%	(Veldkamp and Keller 1997)
Clayey (fine)	1.26%	(Veldkamp and Keller 1997)
Sandy (coarse)	0.78%	(Marquina et al. 2013)

On the other hand, for the estimation of the N_2O indirect emissions two flows were considered: NH_3 volatilization and NO_3 leaching. For the former, IPCC proposes $Frac_{GASF}$ (Fraction of synthetic fertilizer N that volatilizes as NH_3 and NOx) and $Frac_{GASM}$ (Fraction of applied organic N fertilizer that volatilizes as NH_3 and NOx) values of 0.1 and 0.2, respectively, but higher figures (0.15 and 0.30), extracted from Bouwman et al. (2001) for Central America and upland crops, were used here instead. For the latter, the $Frac_{LEACH}$ (Fraction of all N added to/mineralized in managed soils that is lost through leaching and runoff) default value of 0.3 (for irrigated crops) was used, since all studied plantations have irrigation systems.

2.3.2. Adjustment of emission factors for transports

Some modifications have been made in Ecoinvent emission factors to adapt transport emissions to the real conditions of the study. These affect mainly the load factor or each vehicle and the use of refrigeration systems, both for terrestrial and sea transport.

Ecoinvent operation emission factors for road transport consider full outbound and empty return journeys (Spielmann et al. 2007). For different situations, a correction was made based on Swiss emission factors that report full truck emission factors (outbound journey) and empty truck emission factors (return leg) separately. Thus, when load factors for outbound and return trips were different than 100 and 0%, respectively, Equation 1 was used to model each of the trips.

$$EF = E \cdot (0.002 \cdot X + 0.4) \qquad \text{Eq. 1}$$

Being EF the new emission factor, E the Ecoinvent emission factor for the operation of the type of truck considered and X the load capacity percentage (0-100).

Ecoinvent database does not include refrigerated transport, so adjustments were required in order to include the inputs and outputs associated to the cooling system. Two types of refrigerated trucks were considered: refrigerated container trucks (T2 and T3) and refrigerated conventional trucks (T5).

In container trucks, goods are cooled by a diesel generator set, and therefore the vehicle fuel consumption increases. Based on the specifications provided by a manufacturer (GTL Reefer 2013), this increase was estimated in 5.15 liters of diesel per hour. Taking into account the maximum speed for heavy vehicles allowed on Ecuadorian highways (Ecuador-vial 2013), an average speed of 60 km/h was considered and therefore the increase in diesel consumption to be added at the Ecoinvent process was 0.086 l/km, together with the associated combustion emissions. The new emission factors obtained are shown in Table 3. Note that refrigerant losses were not included here due to the lack of information.

Transport from the destination port to the ripening center (T5) is done by refrigerated trucks, but not container trucks, so adjustments are also required here. Values for the increase of fuel consumption due to cooling of between 2 and 4 l/h were obtained from literature (Tassou et al. 2009; Ziegler et al. 2013) depending on the truck load and cooling temperature (-20°C to 0°C). Temperature required by bananas is higher (14°C), so the lowest value of the available ones was used: 2 l/h. An average speed of 80 km/h was used for heavy vehicles on French and Spanish highways, and therefore an increase in fuel consumption of 0.025 l/km was obtained.

Refrigerant leakage, considering both its direct emission to the environment and the emissions associated to its replacement, was included here: a trailer requires an average amount of 6.5 kg of refrigerant (Ziegler et al. 2013) and 8% losses were considered every year. R134a was selected as it is one of the most used refrigerants for these vehicles. Assuming that a truck travels 100,000 km/year, a leakage of 0.0052 g R134a/km was obtained. The new emission factors for this type of truck are shown in Table 3.

Table 3. Modification of the emission factors (EF) for road transport in refrigerated trucks.

Ecoinvent process	**Ecoinvent original EF**	**Recalculated EF (Refrigerated container trucks)**	**Recalculated EF (Refrigerated trucks)**
Operation, lorry >32 T, EURO 3	1.15 kg CO_2e/km	1.64 kg CO_2e/km	1.23 kg CO_2e/km
Transport, lorry >32 T, EURO 3	0.121 kg CO_2e/tkm	0.163 kg CO_2e/tkm	0.128 kg CO_2e/tkm

Regarding sea transport, Ecoinvent provides an emission factor for vessel operation which is not consistent with the actual conditions of the ships used in banana maritime transport (Table 4).

Ecoinvent dataset considers an empty return of the ships to their origin, but according to data provided by the shipping companies, 20% of their capacity is used in the return trip, which is uncooled, and therefore only 80% of that journey has to be allocated to the banana value chain. Applying the equation obtained for road transport, the resulting factor would be 95.2% of the original emission factor.

Concerning the additional fuel consumption for refrigeration, a TEU container has an electric power of about 4.1 kW (4 reefer 2013), which combined with the ship's maximum speed (14 knots) results in 0.158 kWh/km/TEU. Considering 160 g fuel/kWh (Ziegler et al. 2013), an additional consumption of 25,32 g fuel/km/TEU or 2,91 g fuel/tkm (1 TEU = 8,71 ton banana) is obtained. . Regarding refrigerant leakage, 0.88 gCO_2e/tkm was used (Luske 2010).

Table 4. Modification of the emission factors (EF) for sea transport in refrigerated container ships.

Ecoinvent process	**Ecoinvent original EF**	**Recalculated EF**
Operation, transoceanic freight ship	0.009 kg CO_2e/tkm	0.0199 kg CO_2e/tkm
Transport, transoceanic freight ship	0.011 kg CO_2e/tkm	0.0216 kg CO_2e/tkm

2.3.3. Adjustment of emission factors for electricity

Ecoinvent database provides emission factors for individual sources, such as coal or hydropower, as well as the mix for several countries but not Ecuador. Therefore, the national electricity production profile for 2010 was defined (IEA 2010) and the corresponding emission factors were calculated. Moreover, the Spanish electricity mix already included in Ecoinvent was updated by adapting it to the current distribution of sources (REE 2012).

3. Results

3.1. Carbon footprint of the Ecuadorian banana supply chain

According to the results obtained (Table 6), the carbon footprint of Ecuadorian banana consumed in Spain is 1.25 ton CO_2e/ton banana or 0.84 ton CO_2e/ton banana if consumption is excluded and therefore the system ends up at the RDC. The stages that contribute most are: Plantation (S1: 22.1% on average), Consumption (S8: 19.2% on average) and Maritime Transport (T4: 18.7%). Differences are located on the stages placed in Ecuador (S1-S2 and T1-T2-T3) as once the fruit is shipped the downstream processes are exactly the same to all.

Table 6. Carbon footprint results of the average banana value chain, for each plantation type.

Plantation type	**Total**	**S1[a]**	**S2[a]**	**S3[a]**	**S4[a]**	**S5[a]**	**S6[a]**	**S7[a]**	**S8[a]**	**T1[a]**	**T2[a]**	**T3[a]**	**T4[a]**	**T5[a]**	**T6[a]**	**T7[a]**	**T8[a]**
Organic average	1.23	0.26	0.01	0.00	0.00	0.03	0.00	0.06	0.24	0.00	0.01	0.00	0.23	0.23	0.04	0.12	0.00
Conventional average	1.29	0.32	0.01	0.00	0.00	0.03	0.00	0.06	0.24	0.00	0.01	0.01	0.23	0.23	0.04	0.12	0.00
Global average	1.25	0.28	0.01	0.00	0.00	0.03	0.00	0.06	0.24	0.00	0.01	0.01	0.23	0.23	0.04	0.12	0.00

[a]S1: Farm; S2: Collection center; S3: Departure port; S4: Destination port; S5: Ripening; S6: Regional Distribution Center (RDC); S7: Retail store; S8: Households; T1: Transport from farm to collection center; T2: From collection center to departure port; T3: From farm to departure port; T4: Overseas transport; T5: Destination port to ripening; T6: Ripening to RDC; T7: RDC to retail store; T8: Retail store to households.

As explained earlier, farms inventoried in this study were classified according to their production system (organic and conventional). A significant difference was found between the average organic (0.25 ton CO_2e/ton banana at the farm gate) and conventional plantation (0.31 ton CO_2e/ton banana at the farm gate).

3.2. Excel CF calculation tool

An Excel tool for the automatic calculation of the carbon footprint was developed in order to allow future users to obtain an independent estimation of the CF value, from a set of input data (Hospido 2014). This tool allows the user to change a significant number of input parameters related to the plantation stage (S1) and the Ecuadorian transport stages (T1, T2 and T3), analogous to those requested in the questionnaires sent to farmers.

In addition, the spreadsheet includes default values for the rest of the systems, which can be modified in case of having better quantifications or different destinations.

Once all data is introduced, the tool displays a report that summarizes the main results of the calculation, which can be exported as a pdf file. These CF results are divided into subsystems, and major contributions to the total footprint are identified for the most relevant ones. In addition, a comparison is shown among the CF of the actual banana value chain, the average value obtained in this study for Ecuadorian plantations, and the values obtained by other authors for the carbon footprint of bananas and other fruits.

4. Discussion

Figure 2 presents the comparative results of this study with those reported by other authors (Eitner et al. 2012; Iriarte et al. 2014; Lescot 2012; Luske 2010; Svanes and Aronsson 2013).

Luske (2010) carried out a cradle-to-gate analysis of Costa Rican bananas to be consumed in Germany (the chain ends at the retailer). Svanes and Aronsson (2013) is based on the previous study, but consumption is now placed in Norway and results are reported both for cradle-to-gate (up to the retailer, Svanes I column in Figure 2) and cradle-to-grave (i.e. including banana consumption, Svanes II column). Eitner et al. (2012) evaluated five banana plantations in three different Latin American countries, in a study that goes till the RDC assessing different locations in Europe. Lescot (2012) compiles four case studies from three different farms: Plantation A corresponds to Luske (2010), while results included in Figure 2 start from farms B and C, ending up either at the RDC (Lescot BI) or at the European harbor (Lescot BII and C). Finally, Iriarte et al. (2014) presents two results, both ending at the European port, which differ only in the overseas transport stage: a best-case-scenario (Iriarte I) that considers overseas transport in container vessels, and a worst-case-scenario (Iriarte II) in which small reefer ships are assumed.

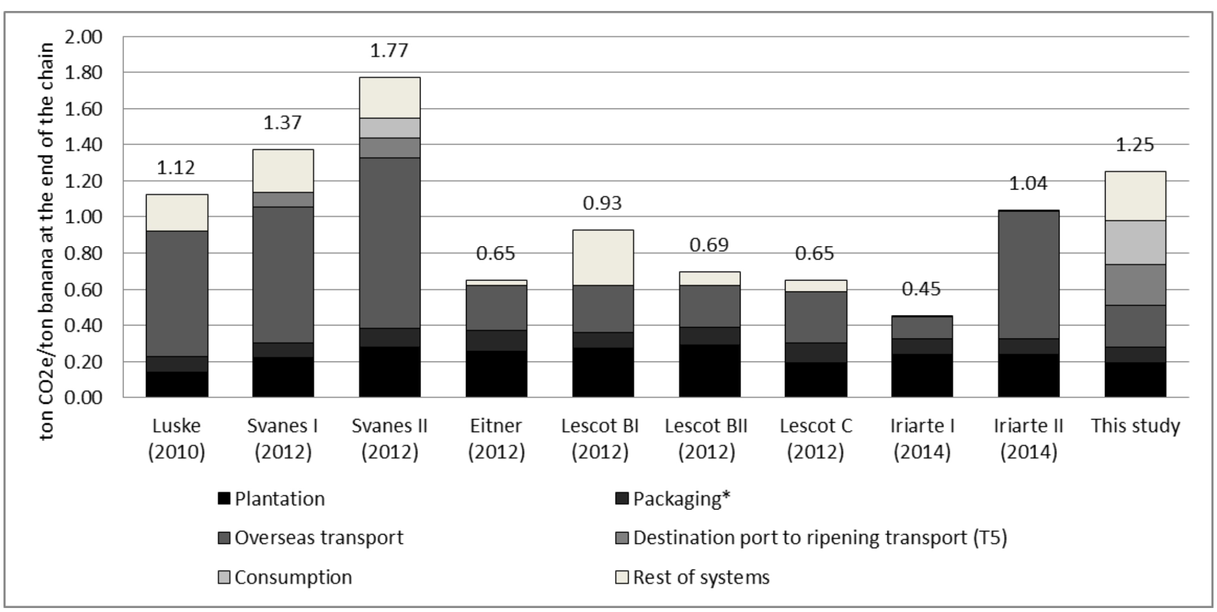

Figure 2. Comparison of the results obtained in this study with those reported by several authors.
* Packaging at plantations is separately reported here in line with the other studies on banana value chain.

The results obtained in this study are lower than those reported by Svanes and Aronsson (2013) for the whole value chain: 1.25 versus 1.77 ton CO_2e/ton banana; and also lower than the figures reported by Luske (2010) and Svanes and Aronsson (2013) until the retailer: 1.01 versus 1.12 and 1.37, respectively, ton CO_2e/ton banana. The main reason behind this difference is the smaller contribution of shipping reported here, due to i) the bigger size of the vessels considered (deadweight tonnage : 40,000 versus 15,000 tons) so the fuel usage per unit of banana transported is lower, ii) the fact that return trip is partially used (20%) for transporting other goods as reported by the shipping companies (while the other studies assumed an empty return) and iii) the assumption of efficient use of the vessel capacity (100% according also to the information provided by the shipping companies), different to the 65% reported by Luske (2010).

Our results are also lower than the ones reported by Lescot (2012) for the chain up to the RDC: 0.84 (this study) versus 0.93 ton CO_2e/ton banana, but higher than those reported by Eitner et al. (2012): 0.65 ton CO_2e/ton banana at the RDC. In both cases the reasons behind the differences are hidden behind the lack of information provided due to confidentiality issues by the former and due to the use of black-box software by the latter.

Results obtained in this study up to the destination port, 0.54 ton CO_2e/ton banana, are lower than the ones reported by Lescot (2012), 0.65 and 0.69 ton CO_2e/ton banana at the European port, and between both values provided by Iriarte et al. (2014) for the best and worst scenarios: 0.45 and 1.04, respectively. Lescot (2012) does not reveal the characteristics of any of the stages included in his assessment, while Iriarte et al. (2014) consider a best-case scenario, in which a non-refrigerated Ecoinvent process is directly used to model overseas transport, and a worst-case-scenario in which emissions from sea transport are based on data from Luske (2010).

5. Conclusion

This study has calculated the carbon footprint associated to banana produced in Ecuador and consumed in Spain (Europe). The whole value chain has been covered, with special emphasis on the activities that take place in the banana plantations: fruit cultivation and packaging. To do so, two groups of farms were assessed based on their production system and significant differences were found between the organic (0.25 ton CO_2e/ton banana) and the conventional plantations (0.31 ton CO_2e/ton banana).

Looking at the whole value chain, the carbon footprint of Ecuadorian banana consumed in Spain is 1.25 ton CO_2e/ton banana or 0.84 ton CO_2e/ton banana leaving the RDC. The stages that contribute most are: Plantation (22.1%), Consumption (19.2%) and Maritime Transport (18.7%).

Taking into account the previous studies on banana carbon footprint and the uncertainties and weakness there identified, special effort has been made here in refining the calculation of N_2O direct emissions at plantation and maritime transport. The values reported in this study are lower or in line with those available in the literature but this calculation is considered more adapted to the characteristics of the actual case study: the Ecuadorian banana.

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LCA applied to sustainable diets: Double Pyramid and Tool Chef to promote healthy and environmentally sustainable consumption

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ABSTRACT

Barilla Center for Food and Nutrition created the Double Pyramid as a means of communicating that foods we should consume often for our wellbeing tend to yield least environmental impacts. From this consideration it was created a tool that calculate both nutritional values (in terms of carbohydrates, protein, fats and energy content) and environmental impacts of the foods commonly eaten (using LCA approach). Results show that a meat-based menu has an environmental impact that is two and a half times higher than the vegetarian one. Based on this data, we can hypothesize how impactful could be simple changes of an individual eating habits' on the environment. Taking the example of a weekly human need of food, in line with nutritional recommendations, we can hypothesize that limiting animal protein to just twice a week can "save" up to 20 square global meters per day.

Keywords: environmental impacts of food, education, nutrition, sustainable diets

1. Introduction

Nowadays it is well-known that proper nutrition is an essential condition for health (Must et al., 1999; Burton et al., 1985). At the same time, in recent years we are facing a growing awareness regarding the environmental consequences of our actions towards the environment (Wackernagel et al., 1996). And in particular, regarding the environmental impacts of food production (Sonesson et al., 2009; Williams et al., 2006).

Since 1992, the Mediterranean Diet has been represented in many documents with the scheme of the Pyramid (Keys et al., 1967; Keys et al. 1980). This graphic form makes it possible to highlight the fact that the basis of nutrition consists of foods of plant origin, typical of Mediterranean eating habits, rich in nutrients (vitamins, minerals, water) and protective compounds (fibres and bioactive compounds of vegetable origin), and narrows towards the top to suggest a less frequent consumption of foods with increasing energy density, usually made from animal protein, fat and simple sugars. After more than 50 years of studies and research, the Mediterranean Diet has been recognized by UNESCO as an intangible heritage of humanity.

The value of the Food Pyramid is twofold: on the one hand is an excellent summary of the main knowledge in studies on nutrition, essential for anyone who pays attention to their health, on the other hand is a powerful tool for education and consumer through to its simple and intuitive schematic diagram.

Many studies have shown that the Mediterranean diet yields positive effects on both health and environment. On the basis of such premise, Barilla Center for Food & Nutrition (BCFN) has closely examined the relation between the nutritional and environmental aspects of food and, in 2010, decided to re-propose the traditional food pyramid model, which was elaborated and updated to carefully integrate the latest findings on nutrition research, combined with the impact of food on the environment and global warming. Combining the two pyramids results in a clear coincidence of a single dietary pattern of two different but equally important objectives: health and environmental protection. This is the BCFN Double Pyramid (see Figure 1) that combines the traditional food pyramid model with the environmental impacts of food (BCFN, 2009, 2010, 2012).

The environmental part of the double pyramid was instead designed by BCFN reclassifying food no longer in function of the nutritional characteristics rather regard to impact on the environment: using data from impact per kilo of product yields an inverted pyramid, which sees foods with greater environmental impact at top and those with reduced impact on the bottom.

The BCFN collected all available and public scientific data to reclassify different kinds of food. Aside from impact on health, reclassification also accounted for food's environmental impact. Use of the Life Cycle Assessment method places all environmental markers on the same level of analysis: carbon, water and ecological footprints were examined as key performance indicators of food production. The current edition engrosses over

1,100 public sources of scientific data. It is necessary to specify that the Double Pyramid has been composed in relation solely to the Ecological Footprint.

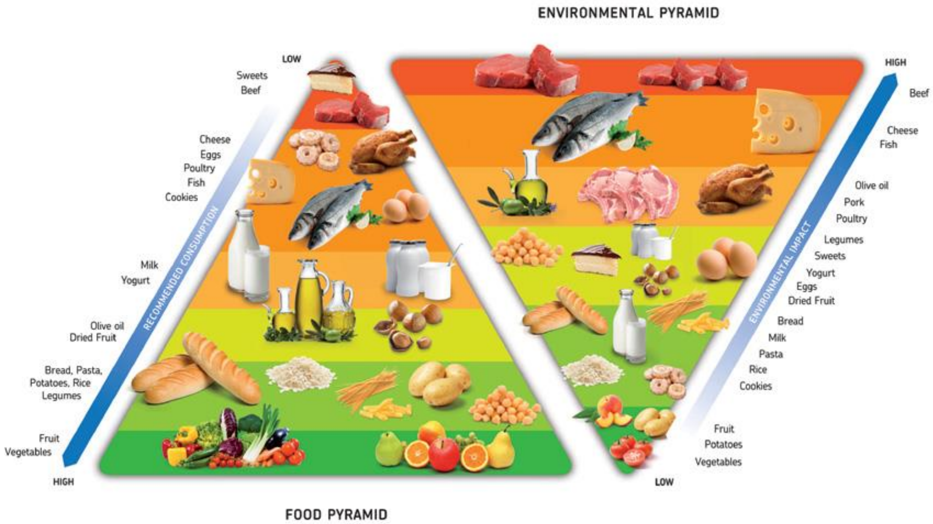

Figure 1. The Double Pyramid [BCNF, 2013]

2. Methods

Starting from the second edition of the Double Pyramid, the BCFN wanted to take a further step, trying to identify the most efficient ways to spread the culture of sustainable diet. BCFN proposes the Double Pyramid as a means of communicating that foods highly recommended by the Nutritional Guidelines in terms of greater consumption tend to yield least environmental impacts. From this consideration it was created a tool that calculate both nutritional values (in terms of carbohydrates, protein, fats and energy content) and environmental impacts of the foods commonly eaten using LCA approach (Andersson, 2000; Baroni, 2006).

The calculation tool, thought to be used by Barilla chefs in different ways, is not aimed at giving nutritional advices from a medical point of view but instead being a cue to spread the proper sensibility towards the right diet and the environment, reconciling human well-being with that of the environment. The purpose of the tool is calculating the environmental and nutritional impacts of dishes and menus.

The database used for the calculation contains specific nutritional and environmental values of more than 250 ingredients. The biggest part of this data derive from the Double Pyramid Database. Also, scientific studies and the Data Banks (Ecoinvent; LCA Food most of all, but also from Environmental Product Declaration Database; Ewig et al., 2006; Foster et al., 2006) are significant data sources and, for some food typologies, as meat, represent the most important source.

As you can see in Figure 2, after choosing food quantity and type and cooking typology and time, Tool Chef gives as output the results in term of nutritional values and environmental indicators. The Environmental impacts of the recipe take into consideration both the impacts of ingredient and cooking phase. That is, the total environmental impacts of the recipe is a sum of the grams of ingredient and the impacts of the ingredient cooked.

	Name or Typology		Tool Chef Database		Total
Ingredients	n grams of FOOD	X	Impact per grams of ingredient	=	Impacts per n grams of ingredient
					+
Cooking	COOKING TYPOLOGY and TIME (X minutes) per grams of FOOD to cook	X	Impact per minute of choosen cooking	=	Impacts per n grams of ingredient cooked per X minutes
					=
				Results	Environmental impacts of the recipe

Figure 2. Tool Chef calculation process

In order to estimate the extent to which the food choices of individuals affect the environmental impacts, two different daily menus were analyzed; in the firs one the protein is of plant origin (Table 1, Vegetarian menu), while in the second, it is mainly of animal origin (Table, 2 Meat-based menu).

Table 1. Vegetarian menu

Breakfast	Mid-morning snack	Lunch	Snack	Dinner
1 portion of fruit 4 rusks	1 portion low-fat yogurt 1 fruit	1 portion of pasta with fennel 1 portion of squash and leeks quiche	1 portion of fruit 1 packet of unsalted crackers	1 portion of vegeta-bles: steamed green beans and potatoes with grated cheese

Table 2. Meat-based menu

Breakfast	Mid-morning snack	Lunch	Snack	Dinner
1 cup of low-fat milk 4 cookies	1 portion of fruit	1 portion of cheese pizza, mixed green salad	1 portion low-fat yogurt 1 packet of unsalted crackers	1 portion of vegeta-bles soup/pasta with peas 1 grilled beef steak 1 slice of bread

3. Results

As you can see in table 3, in order to be properly compared, both menu are balanced from a nutritional point of view and have approximately the same amount of Kcal.

Table 3. Menus nutritional information

	Kcal Total	Protein	Fats	Carbohydrates
Vegetarian Menu	2,030	14%	30%	56%
Meat-based Menu	2,140	15%	25%	60%

Based on Tables 4 and 5, results shown that the vegetable menu has an Ecological Footprint of 19 sq global m^2, while the meat-based one 42. Carbon footprint of the vegetable menu amount at 2,177 gCO_2 eq. versus 7,058 of the meat one. Nearly the same proportion affect the Water Footprint indicator: 2,225 liters for the vegetarian menu versus 5,031 of the meat-based one.

Table 4. Vegetarian menu impact

	Breakfast	Mid-morning snack	Lunch	Snack	Dinner	TOTAL
Ecological Footprint [global sq m2]	1	2	5	1	10	19
Carbon Footprint [g CO2-eq]	150	242	914	108	763	2,177
Water Footprint [liters]	230	242	499	164	1,089	2,225

Table 5. Meat-based menu impact

	Breakfast	Mid-morning snack	Lunch	Snack	Dinner	TOTAL
Ecological Footprint [global sq m2]	3	1	17	2	21	42
Carbon Footprint [g CO2-eq]	270	96	2,963	194	3,535	7,058
Water Footprint [liters]	230	186	1,915	149	2,552	5,031

4. Discussion

Results show that the meat-based menu has an environmental impact that is three times higher than the vegetarian one, which represents a very significant share in the daily environmental impact of an individual.

In table 6 we made estimation on how individual's eating choices can impact on the environment. In particular, we saw that already a small change in eating habits can make the difference. That is, passing from eating meat seven times a week to eating meet twice a week can halve Carbon Footprint (from about 49,000 g CO2-eq to 25,000) and nearly the same for Ecological Footprint (294 global sq m2 versus 179).

Table 6. Variations in the environmental impact depending on eating choices

	Weekly Impact			Average daily impact		
	Carbon Footprint [g CO2-eq]	Water Footprint [liters]	Ecological Footprint [global sq m2]	Carbon Footprint [g CO2-eq]	Water Footprint [liters]	Ecological Footprint [global sq m2]
7 Times Meat Menu	49,406	35,217	294	7,058	5,031	42
5 Times Vegetarian Menu + 2 Time Meat Menu	25,001	21,187	179	3,572	3,027	26
7 Times Vegetarian Menu	15,239	15,575	133	2,177	2,225	19

5. Conclusion

Based on this data, we can hypothesize how impactful could be simple changes of an individual eating habits' on the environment.

Taking the example of a week's human need of food, in line with the recommendations of nutritionists, we can hypothesize that limiting animal protein to just twice a week, can "save" up to 20 square global meters per day.

We believe that with the help of targeted educational campaigns, social marketer can see positive results and shifts in population's eating habits without necessarily compromise their traditions and heritage.

Together with Barilla Center for Food and Nutrition there are many other Institutions and NGO's involved in changing and improving individual's eating choice. Firs of all the FAO with the concept of Sustainable Diets (FAO, 2012). Then another important one is LiveWell for LIFE, an Initiative started by WWF UK and funded by the EU's LIFE Programme for the Environment, that seeks to promote low-carbon and healthy diets in Europe (WWF, 2011).

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Assessment of Biodiversity within the Holistic Sustainability Evaluation Method of AgBalance

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ABSTRACT

Biodiversity cannot scientifically be quantified in its totality. Absolute figures are highly variable. Some categories as systematic taxa etc. are not well established, and the functions and organisms depend on regional and local conditions. Therefore, any quantification of "biodiversity" is an approximation, requiring the relevant elements of biodiversity to be defined and the appropriate indicators used.

In AgBalance, the impact of agricultural activity on biodiversity is assessed as a relative function, constructed from the Biodiversity State Indicator and further indicators that have the potential to increase or decrease biodiversity. In AgBalance, a state indicator is included which allows a widely accepted quantification of the national or regional state of biodiversity. Changes of the state indicator due to influences which come from "driving force" or "response" category indicators provide a valuable tool to assess trends in the development of biodiversity".

Keywords: holistic LCA, biodiversity, AgBalance, sustainability

1. Introduction

The term "biodiversity" was in a widely accepted form defined at the Convention on Biological Diversity (CBD) 1992 as follows: "Biological diversity" means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". According to this definition, biodiversity is encompassing all kinds of life forms, including genetic diversity and the levels of interaction between organisms including the ecosystems which they inhabit. This definition addresses issues that might not seem apparent at first glance. It raises problems on two levels, the level of complexity and data access, as well as on the normative level.

One element of the complexity is the huge range of living organisms and our limited knowledge about them – for example, the number of animal species in the world is estimated to be anywhere between 3 and 30 million. (WWF global, http://wwf.panda.org/about_our_earth/all_publications/living_planet): This is before gene diversity is added to the mix which factors in the huge variability within species and between individuals and populations (TEEB 2010). In addition tiny plants and microorganisms, which are not well known or hardly visible, are usually not included in biodiversity assessments (Reidsma 2006) (SRU 2009). Given the fact that they are responsible for delivering critically important ecosystem services of outstanding importance, like the creation of humus, CO_2-fixation and water balance, this omission is deeply flawed (Burel 1998) (European Union, 2011).

Finally, the quantification of biodiversity depends on a spatial and timely scale (Laurie 2010). The count of organisms under a footstep, in a field, a landscape, or in a larger spatial dimension will certainly result in different figures and data categories.

On the conceptual level, there are several challenges again. Mostly, debates on biodiversity include normative elements (Sarkar 2008) (Henle 2008) (Burkhard 2012). This is evident when studies focus on desirable organisms (in the sense of beneficial or attractive, rare or endangered species). Looking at ecosystem services, the concept of "benefit" implies that human interests are paramount. But, unwanted organisms like parasites, vectors and pathogens are all part of biodiversity, too, which includes organisms that cause damage in agriculture (FAO 2013) (FAO 2010). Animal pests, weeds and plant pathogenic fungi, bacteria, and viruses impact agriculture and reducing worldwide yields. The reality is that these organisms are hugely relevant to the economic, ecological and social pillar of sustainability (Charles 2010) (Karasakal 2009). The complex challenges which are part of the quantification of biodiversity can be summarized as shown in table1.

Table 1. Areas of normative tension when quantifying biodiversity

1. Focus on attractive animals or plants	Versus	Small organisms with often economic and ecological relevance are considered
2. Value-free quantification of biodiversity	Versus	Selection of positively or negatively valued elements of biodiversity
3. Quantification of biodiversity based on complete data sets	Versus	Extrapolation from a subset of data to the totality of biodiversity
4. Numerical quantification of biodiversity	Versus	Semi-quantitative or descriptive characterization of biodiversity
5. Quantification of biodiversity on a small-scale (regional, local, farm) level	Versus	Quantification of biodiversity in general or on a large-scale (national, EU, global) level

The sustainability evaluation method for agriculture and food supply chains "AgBalance™" is based on the principles of life cycle assessments which evaluate impacts which may contain conflicting objectives. For example, a definite farming activity may be positively rated for productivity, but it may be negatively rated for biodiversity development. The intention of AgBalance is to specify alternatives which allow decision makers to prioritize farm management strategies and to specify the respective strengths and weaknesses. Herewith, AgBalance follows the principles of the multi-attribute value theory (MAVT) which allows assessments on the basis of conflicting objectives (v. Witzke 2011).

The need to describe quantitatively sustainability (or elements of sustainability which includes biodiversity) has led to develop AgBalance which is based on precursor systems. In building on the environmental impacts and economic costs assessed in the Eco-Efficiency Analysis (Saling 2005) and the additionally integrated social impact indicators in SEEBALANCE® (Schmidt 2004) (Kölsch 2008), AgBalance™ was designed as a specific type of a new LCA system to assess the sustainability performance of the production of agricultural goods. Therefore, AgBalance in addition to SEEBALANCE contains a range of new agriculture-specific indicators, namely biodiversity, soil health and land use which were identified and developed in a dialogue with various stakeholders. This holistic system allows assessing parameters which contribute to the development of sustainability in agriculture at several user categories:

(1) for the farmers, by assessing current practices and developing scenarios for improved processes,

(2) for the agri-food value chain, by assessing agriculture's contribution over the complete product life cycle and developing options for improvement, and

(3) for policy makers, by assessing the impact of legal bodies and regulations on products and farming practices. Depending on the level of assessment of a study, a "balance" between unequally or even controversially directed factors can be evaluated and assessed by the indicator system.

There are well established systems on the farm level like REPRO (Christen 2009) or KSNL (Breitschuh 2008), and there are systems which support primarily political decisions like SEBI (EEA 2012) or IRENA (EEA 2005). AgBalance was designed to analyze agricultural production including the industrial processes which are described since SEEBALANCE is an integral part of AgBalance. The method focuses on the supply chain level i.e. the crop respectively the cropping system. Indicators are preferably included which can be actively influenced by farmers, for instance properties of agricultural production processes.

In the context of AgBalance biodiversity indicators, several aspects need to be considered. What are the reasons for the selection of definite parameters as indicators, and the disregard of parameters which may be elements of other LCA systems? Firstly, the concept of AgBalance indicators requires numerical quantification of algorithms (Schoeneboom et al 2011). Secondly, AgBalance focuses on anthropogenic factors in general, and this concept is applied for biodiversity indicators, too. Thirdly, indicators are "indirect" in the sense that they do not require monitoring data of definite taxa – like birds, plants or mammals –; instead, parameters which can be regarded as accepted as biodiversity influencers are used as indicators. Obviously, feasibility and practicability play an important role for indicator selection, too. This may go on cost of scientific preciseness (which is, for instance, applied in the BioBio indicator system (Herzog, F. et al., 2012)). AgBalance has not the demand to quantify biodiversity in general, and the same is undoubtedly true for biodiversity assessments in other LCA systems (de Souza, M. et al, 2013; Curran, M. et al, 2011; de Baan, L. et al., 2013). Instead, the indicator set addresses a "biodiversity potential" in the sense that anthropogenic factors which relate to farming are quantified which impact biodiversity in regions where they are implemented. These conceptual elements and in addition the link of biodiversity in the overall concept of sustainability are conceptual characteristics of AgBalance and resulted in the selection of the indicators. Examples are described briefly here.

Subsequently with sustainability evaluation tools (Saling 2002) (Saling 2005) like AgBalance™ (Schoeneboom 2011) (Frank 2012) methodology, environmental, societal and economic impacts are assessed independently.

The environmental impact assessment use characterization factors (as in most LCIA methods) with the resulting impacts normalized to arrive at individual impact categories. The biodiversity results are integrated in the final evaluation as a single result. The normalized results for different environmental impact categories are represented as the environmental fingerprint for each alternative. Relative improvement in each impact is represented by smaller values on the respective axes; hence the smaller the fingerprint, the better the relative performance of the corresponding alternative. Finally, all results are expressed as an overall summary figure.

In order to demonstrate the practicability of AgBalance, a study about winter oilseed rape production in Germany was selected for a case study. Recent reports indicate that oilseed rape production has seen a substantial increase in productivity and profitability for the farmers over the last decade (BLE 2007). Whether or not this intensification has been sustainable in total, however, remained unclear as well as segmental calculations for biodiversity (Vié 2009). Therefore, oilseed rape production in Northern Germany from 1998 to 2008 was analyzed using the AgBalance methodology. In particular, the results and implications of applying the biodiversity indicator set of AgBalance will be discussed.

2. Material and Methods

AgBalance is a life-cycle assessment method for value chains focusing on primary agricultural production that integrates environmental, social and economic cost indicators. It is based on mandatory and optional parts of the ISO 14040 and 14044 standards for life cycle assessment. Furthermore, the developments of different working groups such as UNEP/SETAC for social LCA, the SA 8000 and ISO 26000SR standards were considered in the development of the methods. The results from the individual impact categories were aggregated as outlined (Saling 2002). The method received independent assurance of functionality and coherence from DNV Business Assurance, TÜV Süd and the National Sanitation Foundation (NSF).

Specific impacts of agriculture on the environment (biodiversity, soil health and land use etc.), the economy (total costs and profitability etc.), as well as to social (health and safety, gender equality, working conditions, training, fair trade etc.) aspects are assessed. In total up to 200 metrics are considered to evaluate up to 70 indicators in the three dimensions environment, society and economy. AgBalance offers the flexibility to assess production systems with different regional scope, from the farm level up to regional or even national level. Specific algorithms have been developed that can be operated with different databases, such as specific farm data as well as statistical, sectorial data or survey data. Preferably, publicly available data are used in order to give the system a high level of data transparency. AgBalance further allows for scenario analysis demonstrating interdependent variations of impact results as a function of changes to input parameters.

The set of AgBalance biodiversity indicators follows the "driving force – state – response" model which is proposed by OECD to structure the complex relationships between agriculture and biodiversity (OECD 2003). It is closely related to the "pressure – state – response" concept which can be regarded as largely accepted. Accordingly, the "pressure" component compiles factors which affect negatively biodiversity resulting in a decline of the state of biodiversity (what is reflected in a negative numerical factor on the state). The "state" indicator quantifies the status quo of biodiversity. Generally, organisms which are scientifically or politically accepted are selected for this purpose. "Response" indicators reflect activities which are able to promote or conserve biodiversity.

The indicators are linked with each other by calculation factors which are generated with indicator – specific algorithms. Quantitative ranges which are allowed for each indicator are specified in table 2. The selection of factors between 0.5 and 1.5 is based on bird data of different ecological groups (farmland birds, woodland birds, water and wetland birds, seabirds, butterflies) which show trend curves in this range and indicate herewith numerical realism of calculation factors (Bird indicators, BTO 2014). Maps showing bird populations in Europe as basis for the evaluation can be found at European Environmental Agency EEA under data and maps (EEA).

Table 2. Biodiversity indicators of AgBalance according to the pressure-state-response concept

Indicator	**Calculation factor**
Pressure indicators:	
Low cropping diversity	0.5 – 1
Nitrogen surplus	0.5 – 1
Ecotoxicity potential of pesticides	0.5 – 1
High farming intensity	0.5 – 1
Outcrossing potential	0.5 – 1
State indicator:	
IUCN assessment	(0,7)
Response indicators:	
High cropping diversity	1 – 1.5
Agri-environment schemes	1 – 1.5
Protected areas	1 – 1.5
Low farming intensity	1 – 1.5

2.1 Pressure indicators

“Low cropping diversity” is per se a reduction of agro-biodiversity and provides relatively poor plant-based resources to many animals (FAO 2004). In general, “agro-biodiversity” is more than the pure number of crops: according to FAO, “agrobiodiversity encompasses the variety and variability of animals, plants and micro-organisms that are necessary for sustaining key functions of the agro-ecosystems”. The multitude of crops – i.e. the number of crop species, crop varieties, the number of crop rotation programs on a farm or in a region – are an important element of sustained management of biological resources. In AgBalance, the number of crop plants in a rotation was selected as indicator since the advanced standardization and reduction of complexity of crop rotation programs which is often seen as element of farming intensification has a double negative impact on biodiversity: firstly, it is per se a reduction of agro-biodiversity, and it secondly reduces the diversity of food sources for wild and domesticated animals. Accordingly, under-average numbers of crops in crop rotation programs are selected as pressure indicators in AgBalance. Cropping diversity enhancement is a target of European agricultural regulations with respect to biodiversity protection (summary paper IEEP 2014).

“**Nitrogen surplus”** reduces mostly plant diversity and, based on that, in addition diversity of animals. It is important to make clear that nitrogen is not per se negatively assessed in AgBalance: nitrogen fertilizer – it may be natural or synthetic – is essential for plants. In the context of sustainability, the surplus of nitrogen is critical of several reasons. In the biodiversity indicator set, it is not the potential risk for surface water or groundwater which may be primarily a risk for human health at excessive concentrations. For biodiversity, nitrogen surplus was selected as indicator since it deeply influences plant societies in agricultural landscapes; it often causes the reduction of flowering plants and stimulates grasses. This reduction of primary biodiversity has often indirect negative consequences for pollinating or plant-feeding animals, i.e. the higher levels of the food chain. Therefore, an algorithm for nitrogen surplus is chosen as an indicator for low biodiversity in AgBalance. Appropriate nutrient balances are elements of SAFA (FAO 2012) and IRENA (EEA 2005) indicator systems.

“Ecotoxicity potential of pesticides” is an influencing factor on organisms which are exposed to these compounds. Pesticides – or “plant protection products” – are biologically active compounds which are used by farmers to control weeds (herbicides), fungal diseases (fungicides) or animal pests (insecticides or acaricides). The compounds may have a risk potential, i.e. for unwanted side-effects on such organisms which are not the target of the application by the farmer. Although, registrations which are the precondition for any pesticide application require often a specific risk management by farmers, these products can be handled safely. For instance, buffer zones to surface water to protect aquatic organisms, the risk potential of definite products were selected in AgBalance as in indicator. Data for short-term (acute) and long-term (chronic) eco-toxicity of plant protection products on earthworms, honey bees, rodents, birds, water fleas and fish are therefore available from pesticide toxicity databases. This selection of test organisms follows the European regulations (regulation EC No 1107/2009). Very detailed data on these products can be reviewed and extracted from the European Commission and can be used as well for other regions (EU pesticides database).

"High farming intensity" is a descriptive summarizing parameter; there is scientific evidence that high yields are generally negatively correlated with reduced biodiversity (Geiger et al 2010). This indicator may be challenged since it raises some questions. Empirically, there is the trend that "intensive agriculture" and biodiversity reduction often go hand in hand, while farming practices with reduced productivity – like many organic farming practices, extensive or traditional farming practices – in general promote biodiversity. But, some questions remain open. Is lower productivity in one area (for instance, in Europe) translocating problems to other continents from which agricultural goods need to be imported? This indirect land use or in a more negative sense indirect land use change is often discussed and might be addressed more properly in consequential LCA. However, there is so far no common agreement how to do it. On the other hand a question is, if the negative effect which high productivity often has can be compensated by the generation of more diverse landscapes with hedges, trees and zones which are dedicated to biodiversity protection. Although these questions can be regarded as open, it was decided for the AgBalance indicator set to follow the empiric evidence for a negative correlation between farming intensity and biodiversity. The farming intensity is only one effect to assess the biodiversity; other positive effects from modern farming systems can balance them as well in a positive way.

"Outcrossing potential" means the potential of a crop for fertile reproduction with native wild plants and herewith the genetic infiltration of original biocenoses. Although it may be a numerical increase of biodiversity in natural habitats when outcrossing occurs, there may be competition between indigenous biodiversity and such elements of biodiversity which is introduced by humans. The outcrossing potential according to this indicator is crop specific: there are crops with high outcrossing potential due to parameters like pollen transfer potential, number of wild plants of the same biological taxon in the region (which facilitates inbreeding), seed persistence in soil. Accordingly, a ranking of crops is made which reflects in summary the outcrossing potential of a crop species. Outcrossing potential is in particular an element of political decision making and often controversial public debates in the context of genetically modified organisms (summary paper EFSA 2014).

2.2 State indicator

"Farmland bird populations" are often used as indicators of species diversity (or biodiversity in general) since birds are on top of food chains and require for successful reproduction a diversity of biotic resources (like breeding habitats or different food sources).

In AgBalance, the status of biodiversity in a particular region is assessed by the number of species available that feature on the IUCN Red List for that region (IUCN 2008). The state indicator is calculated from the number of Red List species, according to a non-linear function. The function has the following range: the biodiversity state ranges from 1 (optimal) to 0 (worst); the slope is higher for smaller number of species, i.e., the differences are higher between regions with a smaller number of endangered species than those with higher number of endangered species. For Germany, the 632 Red List species were assigned a biodiversity state of 0.7. This definition is based on the published national biodiversity index of Germany. Other regions are assigned values by interpolation with zero Red List species corresponding to a biodiversity state indicator of 1, and 4000 Red List species to an indicator value of 0.1.

2.3 Response indicators

"High cropping diversity" is per se an increase of agro-biodiversity and provides diverse resources to animals in higher levels of the food chain. "High cropping diversity", i.e. over-average number of crops in crop rotation programs as response indicator, responds to "Low cropping diversity" as pressure indicator (see above).

Cropping diversity is accordingly a bi-directional indicator in AgBalance, since high crop numbers are positively rated, low numbers negatively.

"**Agri-environment schemes"** are subsidies can be applied by farmers to increase elements of biodiversity. Although ARSes may be critically judged (Berendse et al 2004), they are regarded as a response indicator due to the positive effects on biodiversity which they have in total on biodiversity (European Commission 2014). AR-Ses are implemented on a local level, therefore, different measures may be funded which promote biodiversity: it may be extensive farming to promote definite birds species or the sowing of flower strips from which pollinating insects benefit. Two factors are considered in this AgBalance indicator algorithm: the level of funding which the farmer receives, and the acreage for which this funding is dedicated. Both figures and regional information as well can be used and quantified as additional indicators for the change of biodiversity after applying different subsidies.

"Protected areas" are a classical tool of nature conservation to increase elements of biodiversity. In general, there are many different categories of nature conservation territories which contribute to the increase of biodiversity**.** Farmland is often part of such protected areas, for instance in Natura 2000 territories which clearly contribute to the enhancement of biodiversity (European Commission 2014). Therefore, the local range of protected land is selected as an AgBalance indicator.

"Low farming intensity" practices describe in summarizing form the trend that low yields often promote elements of biodiversity (European commission 2014, Geiger et al. 2010). In AgBalance, farming intensity is (like cropping diversity) a bi-directional indicator: low farming intensity practices often contribute to biodiversity conservation, while high farming intensity has the opposite result (see above under pressure indicators). High nature value practices which contribute to biodiversity are mostly associated with lower productivity than conventional farming (BFN 2014).

The above mentioned indicators fulfill the requirement for indicators in the sense that they are influencing factors, which can be regarded as publicly and scientifically accepted. But, this does not allow the quantitative assessment of the biodiversity range which is relevant in anthropogenic landscapes. The concept of using bird population trend curves is applied in AgBalance as a data-based approximation for this quantification. In European countries (and globally) it is shown that the range of 0.5-1.5 gives a realistic dimension (Global wild bird index 2010; Bird indicators 2013; BLE 2007). Based on these data, the biodiversity indicators are connected by factors within a range between 0.5-1.0 for driving force indicators, 1.0-1.5 with response indicators, and a state indicator which is normally <1, for Europe it is 0.7 unless national data are available. This factor depends on the region and former losses of biodiversity to a defined status. A desired trend according to this concept is the positive shift of the state indicator; ideally the state indicator should reach 1.

Competition between agricultural landscapes and natural areas is an important factor in the context of global biodiversity development. It is specified in the "environmental pillar" section, as part of the indicator "land use". It is not an element of the set of biodiversity indicators described here, but a standard element of each study since "land use indicators" are obligatorily used.

Each AgBalance study requires the assessment of all indicators in all three pillars – ecology, economy, social. Herewith, it is avoided that a targeted selection of indicators results in an unbalanced or biased assessment. A study result cannot be exclusively driven only by biodiversity or social or soil indicators or any other selected set of indicators even when the study question refers to elements of sustainability. All single results are displayed and shown separately in the study report. Finally they will be aggregated to an overall result.

3. Results and Discussion

For the showcase study on winter oilseed rape production in the Northern German state Mecklenburg-West Pomerania, the main data sources were from the German "reference farm network", the German annual agricultural report and interviews conducted on five farms, each with 100 – 500 ha of agricultural area planted with winter oilseed rape. The general study's scope was the sustainability performance of winter oilseed-rape production in the study region in 2008 in comparison to production ten years before. The customer benefit was defined as the 'production of 1 ton of oilseed rape, cradle to field border, in Mecklenburg-Vorpommern, Germany, product water content below 9%, data from 2008 and 1998.

Biodiversity indicators are shown in table 3. The indicator results of the two alternatives 1998 and 2008 can be compared with numerical results and have been calculated from single impact categories. The indicators

"Ecotoxicity potential", "Nitrogen surplus" and "Protected areas" show quite good improvements in the 10 years period, the others are constant but not negative. Only the Biodiversity state indicator was reduced which can be linked to other agricultural or industrial processes and the reduction of species in that region. Although the absolute number of pesticides which were used in the standard application schemes increased (5 in 1998 comparing to 9 in 2008), the ecotoxicity potential showed a positive trend due to more favorable ecotoxicity profiles of the new products.

Table 3. Overview on biodiversity impact categories for 1998-2008 comparison for the winter oilseed rape production

		2008 Standard	1998 Standard
Biodiversity	Biodiversity state indicator	0,69	0,70
	Agri-environmental schemes	1,00	1,00
	Protected areas	1,28	1,09
	Crop rotation	1,00	1,00
	Ecotox potential	0,91	0,88
	Farming intensity	0,60	0,60
	Intermixing potential	0,65	0,65
	Nitrogen Surplus	0,75	0,69
	Result biodiversity	0,24	0,18

The protected area coverage evaluates the coverage of Natura2000 and FFH areas in the study region (in this case, the federal state) against a given target value, as well as the fraction of area enrolled in agri-environmental schemes.

The results of the evaluation of the Biodiversity Potential can be linked afterwards with results from other environmental indicators to come to conclusions which are easy to understand, consider the whole life cycle and are holistically done. With the final results, further decision-making can be initiated and can be used for further improvements of the agricultural processes.

The highest aggregated assessment of all sustainability indicators showed a substantial improvement of sustainability score throughout all dimensions ("Economy", "Environment" and "Society") when comparing the good agricultural practice "Standard 1998" to "Standard 2008" practice. It is evident that ecological indicators contributed significantly to this difference since they exhibit clear differences between 1998 and 2008. In Figure 1 it is shown, in which environmental indicators, the alternatives differ and where improvements happened during the time period of 10 years. As more outside an alternative is located for the specific indicators, as more this alternative have benefits in the defined categories. The figure shows that in 2008 compared to 1998 all indicators have been improved; just the soil indicator showed the same result.

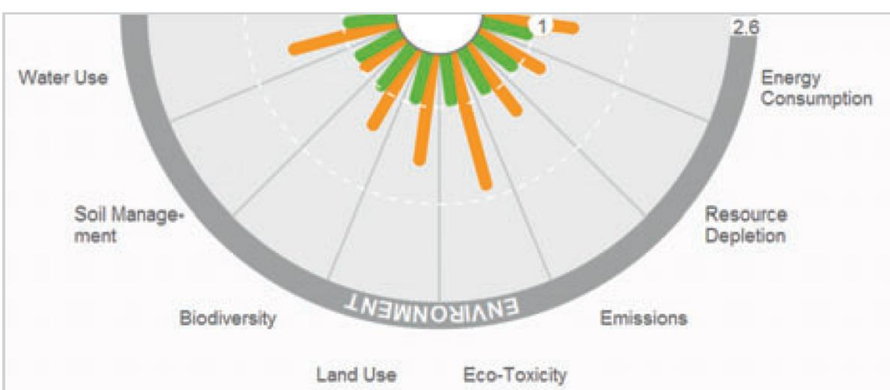

Fig. 1 Assessment results of biodiversity in total in the context of the other indicator sets of the environmental pillar (Green petals reflect the values for 1998, orange petals represent 2008).

The overall results can be shown in a similar way, covering all types of indicators to get a good overview, were improvements have been made over the last 10 years and where further activities or changes needed to have a more sustainable solution. Figure 2 shows all the impacts and allows specialists together with farmers the development of strategies for higher sustainability in the sector. In all main categories, the 2008 technology is more sustainable compared to 1998. It can be shown with the AgBalance methodology and can be quantified, that the sustainability in the defined type of agriculture was improved over the last 10 years. A common opinion in the society that the biodiversity was reduced if the situation today is compared to the situation 10 years ago, can be refuted.

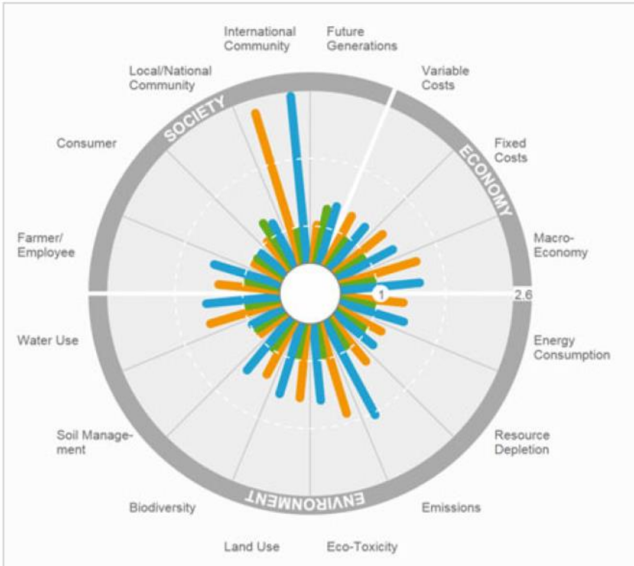

Fig. 2 Assessment results of all indicators in total (Blue petals reflects the values for modified fertilizer regime, based on Good Agricultural Practice in 2008, Green petals reflect the values for 1998, orange petals represent 2008).

4. Conclusion

The indicator set which is used to assess biodiversity in AgBalance has definite properties which become particularly visible with the showcase study. The concept implements that agriculture is not „good“ or „bad“ for biodiversity in general, instead, specific agricultural measures may promote or reduce biodiversity. A “state indicator” is a valid tool to make assumptions about biodiversity in general, based on drawing extrapolations from organisms of a select group to describe the state of biodiversity in total at a definite time and at a definite place. The amount of detailed information on various aspects of the sustainability performance of the production system together with scenario analysis makes AgBalance a powerful tool to derive recommendations for optimized crop production protocols.

Several examples show, that this kind of information is very helpful for farmers and other members of the whole supply chain, to develop more sustainable processes and to support a more sustainable agriculture. The quantification of sustainability indicators show very clearly optimization potentials and can be used for further improvement of processes. It can support the positioning of farmers in the society and increase the acceptance of products in the market following general principles of sustainability as a market trend. In particular the biodiversity facts of an agricultural process are in the focus of different stakeholders and needs to be improved to reach the goals of generating higher state indicators of a certain time period. The methods shown in this article are bale to measure and assess these effects and giving clear directions for the improvement of biodiversity.

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Life Cycle Assessment of French livestock products: Results of the AGRIBALYSE® program

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ABSTRACT

In 2009 two French laws were passed on the provision of reliable and complete environmental information on "product plus packaging" to consumers. The AGRIBALYSE programme has produced an LCI database of agricultural products at farm gate to: i) support environmental labelling and ii) provide benchmarks for improving agricultural production systems. AGRIBALYSE analysed a wide range of animal Product Groups and Contrasted Production Systems. LCIs were calculated using a single methodological frame (Koch and Salou, 2014). The Functional Unit for the study was kg (life weight, Fat and Protein Corrected Milk, egg). The indicators Global Warming Potential, non-renewable fossil energy demand, acidification, eutrophication and land occupation were analysed. This paper presents a synthesis of AGRIBALYSE results with a focus on pig production systems. In the last part of this paper we expressed our results using an economic allocation method to better compare them to literature references.

Keywords: Livestock production, Life Cycle Assessment, LCI database, Review

1. Introduction

In 2009 two French laws were passed on the provision of reliable and complete environmental information on "product plus packaging" to consumers. The Life Cycle Assessment (LCA) method was chosen to assess environmental impacts of goods. ADEME, the French Environment and Energy Management Agency, was mandated to set up a Life Cycle Inventory (LCI) database to support this policy. The AGRIBALYSE program (Colomb et al., 2014) started in 2010, involving: i) ADEME as project coordinator; ii) Agroscope (Switzerland) for plant production and INRA for animal production as project co-leaders; iii) CIRAD, ACTA and 10 technical agricultural institutes as project partners. The aim of AGRIBALYSE was to provide LCIs of French agricultural products at farm gate to: i) support environmental labelling and ii) provide benchmarks for improving agricultural production systems. To ensure consistency between the various products of the database, a general methodological framework for the programme was defined (Koch and Salou, 2014). It was decided that the methodologies used and the deliverables must be consistent with ILCD recommendations (JRC and EIS, 2010). The database contains LCIs for 28 crop and 18 animal products, for many products several LCIs exist, reflecting different production systems. In total 113 LCIs are available in a unit process format (www.ademe.fr/agribalyse-en).

In this paper, we first present the AGRIBALYSE database results for animal products. The second part of the article focuses on the environmental impacts of pig production systems. The third part compares AGRIBALYSE results to literature results for animal product LCAs.

2. Methods

2.1. Choice of systems studied

Within AGRIBALYSE, the choice of systems studied was based on an analysis of agricultural products most consumed in France (BIO IS, 2010). Studied systems were chosen according to two main criteria: i) systems representative of French production, to match the first objective of AGRIBALYSE; ii) contrasted or innovative systems, to match the second objective. For animal production, 44 systems, distributed within18 product categories, have been studied (Koch and Salou, 2014).

2.2. Methodology for LCA

The Functional Units (FUs) retained were kg of live weight, kg of Fat and Protein Corrected Milk (FPCM) and kg of egg. Inventories were calculated from “cradle to farm gate”, so transformation processes were not included. Emissions and resource use for animal productions were calculated according to Koch and Salou (2014). The models used to calculate direct emissions from livestock production are presented in Table 1.

Livestock production systems are often multifunctional and thus a production system frequently produces two or more co-products. Consequently, a method to assign environmental impacts to co-products is needed. ISO 14044 (2006) gives recommendations concerning co-product handling: i) avoid allocation; when allocation is unavoidable, allocate the impacts according to ii) a physical criterion that reflects the underlying relationships between the co-products, or iii) the economic value of each co-product. To match these recommendations, AGRIBALYSE developed a two step “biophysical” approach to handle livestock co-products (Koch and Salou, 2014). In the first step, allocation is avoided by dividing the production system in several unit processes. Each of these corresponds to a characteristic physiological stage of the animal. When a stage yields a single product, all impacts are attributed to this product. Thus for several stages, allocation is avoided. For stages yielding several products, allocation is based on the metabolic energy required to produce each co-product. The metabolic functions considered are: maintenance, activity, growth, lactation, gestation and wool production.

Table 1. Models used in the AGRIBALYSE program to calculate emissions and resource use directly linked to livestock production. CO_2: carbon dioxide; VOC: volatile organic compound; SO_X: sulphur oxide; NO_X: mononitrogen oxides.

Substance emitted / Resource consumed	Source of emissions / consumer of resource	Literature reference for model used
Ammonia (NH_3)	Animal excretion (building/storage)	
	- calculation of nitrogen excreted	CORPEN 1999a, 1999b, 2001, 2003 and 2006
	- emission factors	EMEP/EEA 2009 Tier 2
Combustion gas	CO_2	ecoinvent v2 (Nemecek and Kägi 2007), using an LCI “combustion of diesel/kerosene” data set
	Other air pollutants (metals, VOC, SO_X, NO_X…)	
Methane (CH_4)	Animal excretion (building/storage/grassland/outdoor run)	IPCC 2006 Tier 2
	Emissions from enteric fermentation: cattle and sheep	IPCC 2006 Tier 2
	Emissions from enteric fermentation: other animals	IPCC 2006 Tier 1
Nitrate (NO_3)	Outdoor runs	Basset-Mens et al (2007)
Nitric oxide (NO)	Buildings and storage	EMEP/EEA 2009 Tier 1
Dinitrogen oxide (N_2O)	Buildings and storage	IPCC 2006 Tier 2
Land transformation	All types of production	ecoinvent v2 (Frischknecht et al. 2007)
Phosphorus , nitrogen, total suspended solids (TSS)	Aquaculture	Papatryphon et al. (2005)

Several impact indicators have been retained for AGRIBALYSE (Koch and Salou, 2014), following ILCD recommendations. Indicators such as: GWP_{100} (IPCC 2006); acidification CML 2001 (Guinée et al. 2002); eutrophication CML baseline 2000 2.5 (Guinée et al. 2001); land occupation CML 2001 (Guinée et al. 2002); non renewable energy, fossil + nuclear SALCA (pers. com. SALCA – Swiss Agricultural Life Cycle Assessment) have been studied.

2.3. Comparison with other studies

The selection of the LCA studies for the comparison (Table 5) was based on a modified version of the criteria defined by de Vries and de Boer (2010). Our criteria were: i) studies published in peer-reviewed scientific journals or scientific reports; ii) studies from OCDE countries and partners; iii) studies of systems that produce pig, poultry, beef, milk, egg or fish; iv) attributional LCA studies; v) studies using economic allocation for co-products; vi) "cradle to farm gate" studies; vii) studies published after 2004.

A common FU is needed to compare our results to literature results. When necessary, literature results were recalculated to match AGRIBALYSE FUs. In this article, we focussed the comparison on GWP. To facilitate comparison, AGRIBALYSE results were recalculated using economic allocation instead of biophysical allocation. Economic allocation factors used and sources for economic data are presented in Table 2.

Table 2. Economic allocation factors and sources used to recalculate AGRIBALYSE results.

Products	Sources and allocation factors (%)	Products	Sources and allocation factors (%)
Cow milk	Nguyen et al. (2013)	*Goat milk*	Kanyarushoki et al. (2009)
Milk	86.6	Milk	97
Calf	4.2	Cull goat and kid goat	3
Cull cow	9.2		
Suckler cow	Nguyen et al. (2012)	*Pig*	Basset-Mens and van der Werf (2005)
Cattle weaner	68	Pig	93.5
Cull cow	32	Cull sow	6.5
Ewe milk	IDELE (2011), A. Gac (IDELE) pers. com. (2014)	*Egg*	P. Ponchant (ITAVI) pers. com. (2014)
Milk	79.9	Egg	99
Cull ewe	2.8	Cull hen	1
Lamb	17		
Wool	0.3		

3. Results and discussion

A synthesis of AGRIBALYSE animal product LCA results is presented in Table 3[1] (detailed results available: www.ademe.fr/agribalyse-en). Relative to current LCI databases and LCA results for agricultural products the AGRIBALYSE database presents a major advance, as it contains detailed LCI data for a wide range of agricultural products at farm gate, using a homogeneous methodology. For several products a large number of variants, corresponding to different production systems (13 systems for beef, 6 systems for cow milk; 8 for pigs, 6 for eggs, 9 for poultry), are available. Furthermore, AGRIBALYSE contains some products (goat and sheep milk, rabbit, duck, turkey) for which very little or no LCA results were available so far.

Due to the limited size of this paper we will focus our analysis on results for pig production. Several scenarios for feed supply and overall production system were compared to the reference scenario, which represent an average conventional production of pig in France (Table 4). The impacts of the three "feed" scenarios are quite similar to those of the reference scenario. An increase of land occupation is observed for *Pig, fed rapeseed meal*, because rapeseed meal had a higher land occupation than the raw materials it replaced. A modest increase of all impact categories is observed for the *Pig, fed soybean meal* scenario. The *Pig, on-farm feed supply* scenario requires less energy that the reference scenario, as feed is mainly produced on-farm less transport is required.

Table 3.Summary of AGRIBALYSE LCA results[1] for animal products, using a biophysical approach for co-product handling. Means represent an average of impact values of all inventories calculated for a product category within AGRIBALYSE. Results are expressed per kg FPCM (Fat and Protein Corrected Milk) for milk, kg of live weight for animals, kg of egg. CV: Coefficient of Variation.

			Product category								
			Cattle for beef	**Cow Milk**	**Ewe Milk**	**Goat Milk**	**Pig**	**Egg**	**Poultry**	**Rabbit**	**Fish**
Number of systems studied			13	6	1	1	8	6	9	1	3
GWP	**Mean**	**kg CO2 eq**	11.4	0.9	1.5	0.8	2.5	1.8	2.8	2.3	2.9
	Median	**kg CO2 eq**	11.3	0.8	-	-	2.4	1.7	2.9	-	2.4
	CV	**%**	36.5	14.7	-	-	16.2	17.9	24.4	-	45.1
Acidification	**Mean**	**g SO2 eq**	135.3	8.9	25.1	17.5	40.2	38.8	45.5	14.3	14.1
	Median	**g SO2 eq**	136.7	8.6	-	-	35.8	38	44.7	-	12.4
	CV	**%**	33.5	17.5	-	-	18.7	12.2	26.8	-	31.7
Eutrophication	**Mean**	**g PO4 eq**	45.8	3.8	8.9	5.4	16.9	15.3	19.3	7.1	101.8
	Median	**g PO4 eq**	43.0	3.6	-	-	14.1	15.1	18.9	-	56.7
	CV	**%**	29.7	12.2	-	-	77.5	33.7	45.1	-	80.6
Land occupation	**Mean**	**m2a**	21.3	1.6	4.7	1.6	4.7	3.6	4.8	2.8	1.5
	Median	**m2a**	16.6	1.4	-	-	3.7	3.2	4.3	-	1.6
	CV	**%**	58.6	29.5	-	-	51.6	33.7	37.2	-	16.8
Energy Non renewable fossil + nuclear	**Mean**	**MJ eq**	27.5	2.9	4.9	8.5	16.4	16.8	24.9	23.1	49.6
	Median	**MJ eq**	26.5	2.9	-	-	16.5	16.3	24.3	-	41.2
	CV	**%**	53.5	8.5	-	-	4.2	10.1	24.4	-	32.3

Table 4. AGRIBALYSE LCA results of for pig production systems, calculated using the AGRIBALYSE "biophysical" allocation method.

System	**Scenario**	**GWP**	**Acidification**	**Eutrophication**	**Land occupation**	**Energy**
		kg CO_2 eq	**g SO_2 eq**	**g PO_4^{3-} eq**	**m^2a**	**MJ eq**
Pig, French average, conventional production	Reference	2.40	35.2	13.6	3.41	17.0
Pig, fed rapeseed meal, conventional	Feed	2.36	35.5	14.0	3.78	16.6
Pig, fed soybean meal, conventional	Feed	2.51	35.8	14.1	3.60	17.6
Pig, on-farm feed supply, conventional	Feed	2.33	35.0	13.9	3.49	15.6
Pig, excess slurry treatment, conventional	Production System	2.42	35.9	14.0	3.90	16.6
Pig, pig with outdoor run, Label Rouge quality label	Production System	2.83	40.9	15.3	3.66	16.4
Pig, pasture system, Label Rouge quality label	Production System	2.15	49.5	19.8	5.48	15.4
Pig, organic production	Production System	3.47	54.4	30.5	10.56	16.7

For the comparison of production systems, our results show an increase of land occupation for the scenario *Pig, excess slurry treatment*. The use of feed raw materials with higher land occupation than in the reference scenario, explain this difference. The two *Label Rouge* scenarios and the *Organic* scenario used less energy (respectively 4%, 9%, 2%), because the animals live outdoors or in buildings that are not heated nor mechanically ventilated. The systems *Pig, with outdoor run, Label Rouge* and *Pig, organic* used deep litter in buildings, which favours N_2O emission. Consequently, they presented higher GWP impacts. Acidification, eutrophication and land occupation impacts were higher than the reference for the *Pig, pasture system Label rouge* scenario and for the *Pig, organic* scenario. This results from the presence of pigs on pasture in these systems. For *Pig, organic*, which presented the highest values, this trend was reinforced by the use of organic crops as the raw materials for the animal feed. Organic crops had lower yields than conventional crops and consequently higher land use per kg product, while emission per hectare were similar. For those systems, the technical performances (kg feed per kg animal growth) were also lower, which led to an increase of many impacts.

Once converted using economic allocation, AGRIBALYSE results for GWP were compared to results from the literature (Table 5, Figure 1). For most livestock species AGRIBALYSE results were in the same range of values as literature results, except for pigs and eggs where AGRIBALYSE results presented less variability and were in the lower range of literature values. The variability observed has three major causes.

Table 5. Summary of AGRIBALYSE results[1] and selected studies from the literature for GWP. Results were calculated using economic allocation. Means represent an average of GWP value of all inventories calculated for a product category within AGRIBALYSE or literature. Results are expressed per kg FPCM (Fat and Protein Corrected Milk) for milk, kg of live weight for animals, kg of egg. CV: Coefficient of Variation.
1: Williams et al. (2006); 2: Katajajuuri (2008); 3: Casey and Holden (2006); 4: Basset-Mens et al. (2009); 5: Casey and Holden (2005); 6: Thomassen et al. (2008); 7: Thomassen et al. (2009); 8: Kanyarushoki et al. (2009); 9: Basset-Mens and van der Werf (2005); 10: Mollenhorst et al. (2006); 11: Seguin et al. (2013); 12: Leinonen et al. (2012); 13: Prudencio da Silva (2014); 14: Boissy et al. (2011); 15: Aubin and van der Werf (2009).

Product category	**Number of systems studied**	**GWP**			**Product category**	**Number of systems studied**	**GWP**		
		Mean	**Median**	**CV**			**Mean**	**Median**	**CV**
		kg CO_2 eq		**%**			**kg CO_2 eq**		**%**
Cattle for beef					*Egg*				
AGRIBALYSE	13	9.44	9.84	62.9	AGRIBALYSE	6	2.49	2.50	15.6
Literature[1,3]	7	8.70	8.62	35.9	Literature[1,10,11]	9	3.88	4.30	30.3
Cow Milk					*Poultry*				
AGRIBALYSE	6	1.05	0.97	17.2	AGRIBALYSE	9	2.88	2.96	24.5
Literature[1,4,5,6,7,8]	10	1.28	1.36	12.4	Literature[1,2,11,12,13]	10	2.59	2.30	30.4
Goat Milk					*Fish*				
AGRIBALYSE	1	1.03	1.03	-	AGRIBALYSE	3	2.96	2.41	45.1
Literature[8]	1	1.32	1.32	-	Literature[14,15]	4	2.71	2.51	24.1
Pig									
AGRIBALYSE	8	2.74	2.62	14.5					
Literature[1,9]	6	4.10	4.65	24.6					

The first source of variability is the diversity of production systems. Indeed, both for the literature results and within AGRIBALYSE, many different production systems exist within a product category. This is particularly true for "cattle for beef" where different production systems, but also different types of animals (dairy cull cow, dairy calves, suckler cull cow, suckler cow) are represented.

The second source of variability is the LCIA method used. The GWP results presented here have been calculated according different methods: Houghton et al. (1994) for the study by Thomassen et al. (2008); IPCC 1996 for the studies by Casey and Holden (2005, 2006); IPCC 2001 for the studies by Williams et al. (2006), Katajajuuri (2008), and Thomassen et al. (2009); IPCC 2006 for AGRIBALYSE, Leinonen et al. (2012), Nguyen et al. (2012, 2013) and Prudencio da Silva et al. (2014). This heterogeneity contributes to the variability of the results.

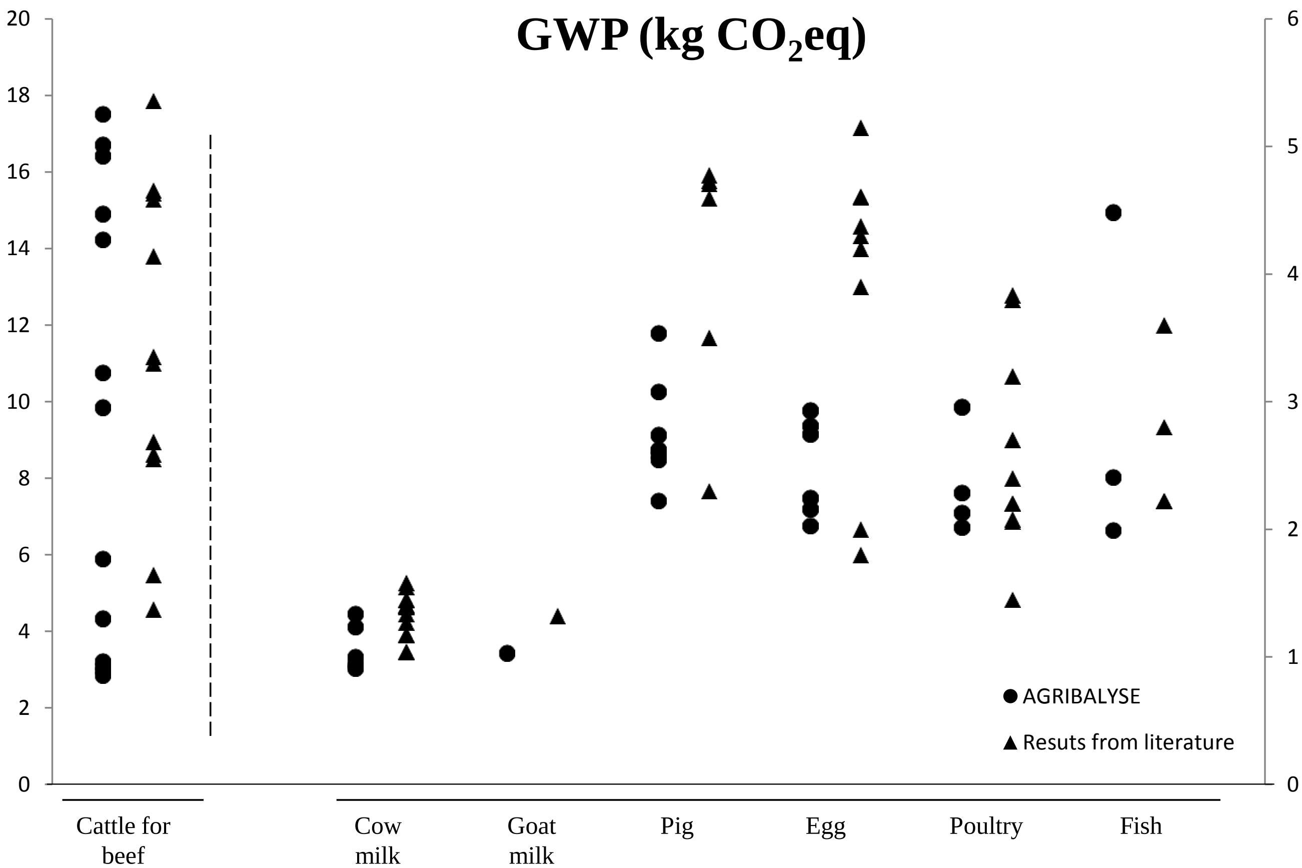

Figure 1. GWP values for products at the farm gate. AGRIBALYSE results[1] are compared to literature references. All results were calculated using economic allocation. Results are expressed per kg FPCM (Fat and Protein Corrected Milk) for milk, kg of live weight for animals, kg of egg. For "Cattle for beef" see left scale, for other products see right scale.

LCI methodological choices are the third source of variability. While all the studies are "cradle to farm gate", many aspects of system modeling vary from one study to another. This particularly concerns the models used to calculate direct emissions, which vary greatly between the selected studies. As an example, N_2O emissions have been calculated according several methodologies depending on studies. Gac et al. (2006, 2010), IPCC (1996, 2000, 2006), UNECE (1999), van der Werf et al. (2009), MfE (2006), among others, have thus been used.

4. Conclusion

AGRIBALYSE provides a large LCI database containing data for French crop and animal products. Its objective is to provide references for a wide range of production systems to support environmental labelling of products and redesign of production systems for improved eco-efficiency. As shown for pig production, the great diversity of production systems of the database combined with AGRIBALYSE methodological framework allows an accurate analysis of farming systems and a transparent comparison of their environmental impacts.

Our results also show that the AGRIBALYSE methodological framework produces results that are largely consistent with literature references, even if differences due to the three sources of variability identified in this article remain.

[1]The results presented here anticipate several corrections that will be taken into account within the version V1.2 of the database. This version V1.2 will be published at the beginning of 2015.

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FoodPrints of Households

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ABSTRACT

We used data from the Swiss household budget survey and applied multiple linear regressions based on generalized linear models to model the at-home consumption of food products and beverages of individual households. Seven household characteristics such as size, income, and educational level served as input variables for the regressions. The food products and beverages demands of 3,238 individual households of a Swiss municipality were environmentally assessed with life cycle assessment. Carbon footprints per household and year vary from 84 kg CO_2-eq. to 4.97 t CO_2-eq. with a mean value of 1,098 kg CO_2-eq. This variability is significantly smaller than the footprint variability for the consumption areas of housing and mobility in Switzerland, where 25% of the people are responsible for 50% of the environmental impacts. Differences between high- and low-impact households can be primarily explained by differences in meat and dairy consumption.

Keywords: Household food impacts, Multiple Linear Regression, Generalized Linear Models

1. Introduction

Together with housing and mobility, food is one of the three environmentally most relevant areas of consumption (Tukker and Jansen, 2006). Households form the smallest organizational units in society and many decisions about food consumption are taken on this level. Understanding the drivers of household food consumption is thus important to design effective policy measures. Existing studies often work with average household demands (Jungbluth et al., 2011) or artificial diets (Dalgaard et al., 2007). Only few studies have considered variability between different household consumption patterns (Girod and De Haan, 2009).

In this study we present a way to model the amount of food products and beverages consumption for individual households on a monthly basis. The model relies on multiple linear regression performed on data of household surveys and is able to predict demand with the help of seven explanatory household parameters.

2. Methods

We used multiple linear regressions to establish relationships between household characteristics and consumption of different food products and beverages. We distinguished between 44 different comestible good categories of unprocessed and pre-processed food products and beverages. The regression was performed with generalized linear models (GLM) using data from the Swiss household budget survey (HBS) (Swiss Federal Statistical Office (SFSO), 2012b).

2.1. Base data

A household budget survey is an inquiry of household financial budgets and expenditures. Such surveys are conducted in all EU states and many other countries worldwide (European Commission, 2003). HBS consist usually of three parts: a record on household variables, a record on household member variables and a revenue and expenditure journal. Information about revenues and expenditures has to be entered by selected households over a certain time period, usually a month. Expenditures are categorized according to the United Nations' "Classification of Individual Consumption according to Purpose" (COICOP) (United Nations Statistics Division 2012). The Swiss version of the HBS is conducted annually from January to December in the seven major regions of Switzerland (i.e. Lake Geneva, Espace Mittelland, North-Western Switzerland, Zurich, Eastern Switzerland, Central Switzerland, and Ticino) and inquires around 10,000 households, of which usually around 30% respond.

2.2. Multivariate regression analysis

Linear regressions were performed for different food product and beverages categories employing GLM and using an iteratively weighted least square algorithm for the maximum likelihood estimation of the parameters. GLM is a unifying approach to generalize linear regression and estimate parameters from multiple linear regressions with dependent variables that are not necessarily normally distributed (Gill, 2001). GLM uses a so-called link function to relate linear regression of independent variables to their dependent response variable. The link functions are chosen according to the distribution of the response variable. Link functions have been formulated for various distributions like binomial, Poisson and normal. The inverse link functions relate the estimated parameters and the explanatory variables to the dependent variable.

Characteristics that we used as independent variables for the regressions were household size (i.e. number of household members), number of rooms in their apartment, number of apartments in their building, age of the oldest household member, and age of the youngest household member. These variables were all given as integer value greater or equal to zero. Monthly household income and highest education of household members were also used. They were given as categorical values. Income is scaled from 1 to 10. Category 1 is equal to 0-1,999 Swiss Francs (CHF), 2 is equal to 2,000 to 2,999 CHF, and so forth. Category 10 is equal to 10,000 CHF or more. Education is also divided in 10 categories ranging from no education (1) to graduate degree (10) (see Supporting Information). These seven explanatory variables were chosen because they can be easily derived from national census data.

The dependent variables of the regressions were the amounts of purchased food products and beverages per household during the reporting period. For the modeling of household demand we pursued a two-tier approach, in which we multiply two multiple linear regressions. The first regression estimates the probability of a good being bought and the second regression determines the bought amount. For instance, vegetarian households are not likely to purchase meat. Some households do not drink wine and pork meat might not be consumed due to religious reasons. Thus, the probability to buy a certain good is higher for some households than for others. We applied multiple logistic regression to dichotomous demand data (i.e. the value was 1 when the purchased amount was greater than zero and 0 otherwise). The multiplication of the probability of a household buying a certain good and the amount of good that will be purchased, if the household buys the good, led to the expected amount of a certain good bought by a household.

The purchase data of one household as reported in the HBS is only collected during one month. Thus, the regressions can only predict the goods demands per households and month. Therefore, we had the choice to either fit regression models for each month of a year or to aggregate the monthly data and fit regression models for an average month of that year. We pursued both possibilities and used always 70% of the applicable data for fitting, 15% for validation and 15% for testing.

The regressions were fitted in a stepwise fashion, meaning that explanatory variables were iteratively eliminated from the regression, if they did not significantly improve the maximum likelihood estimation. Then, the estimated parameters were validated with the corresponding data. Spearman ρ, a rank order correlation coefficient, and the root mean squared error (RMSE) between observations and modeled values were calculated. RMSE describes the standard deviation of the residuals between observed demands $\hat{y}$ and modeled demands.

Each stepwise regression was performed 30 times. After each time the regression was validated. Each time the data for fitting and validation was randomly selected. In the end the best set of estimated parameters was selected according to the combination of the highest ρ and the smallest RMSE.

We performed multiple linear regressions for each possible combination of normal and Poisson distributed response variables and monthly or aggregated data. Finally, the goodness of best set of regression parameters was tested with the testing data. Thereby, RMSE, Spearman ρ and the absolute difference between total observed demand of good and total modeled demand were calculated.

2.3. Case Study

A case study was performed to illustrate the possible application of the household consumption model. The household data for the Swiss municipality Wattwil was drawn from Swiss census data for the year 2000 (Swiss Federal Statistical Office (SFSO), 2012a). Based on the household characteristics from the census, the demands for 44 different food product and beverages categories of 3,238 households in Wattwil (8,075 inhabitants) were quantified, using the outcome of the regression analysis. For each modeled comestible good we derived life cycle impact results for their global warming potential (IPCC, 2007) (GWP, in kg CO_2-eq. per kg comestible good). Most data was taken from ecoinvent 2.2 (ecoinvent Centre, 2012) and the study of Stoessel et al. (Stoessel et al., 2012). This study contains life cycle inventory data from different production countries, thus, we were able to calculate average yearly supply mixes based on foreign trade statistics from the Swiss Federal Customs Administration (Swiss Federal Customs Administration, 2012). For certain comestible goods groups (e.g. meat, fish, and dairy products) we relied on impact results from scientific articles and reports (Alig et al., 2011; Alig et al., 2012; Büsser and Jungbluth, 2009a; Büsser and Jungbluth, 2009b; Eymann, 2012; Humbert et al., 2009; Jungbluth, 2006; Kendall et al., 2010; Martignoni et al., 2012; Mattila et al., 2012; Nemecek et al., 2011; Sonesson et al., 2009; Thrane, 2006).

3. Results and discussion

Figure 1 shows the observed demands of 44 food products and beverages for 3,235 nation-wide surveyed households in the Swiss HBS for the year 2004 (left) and the modeled demands for these households (right). The demands per households were normalized by the number of household members to adjust for different household sizes.

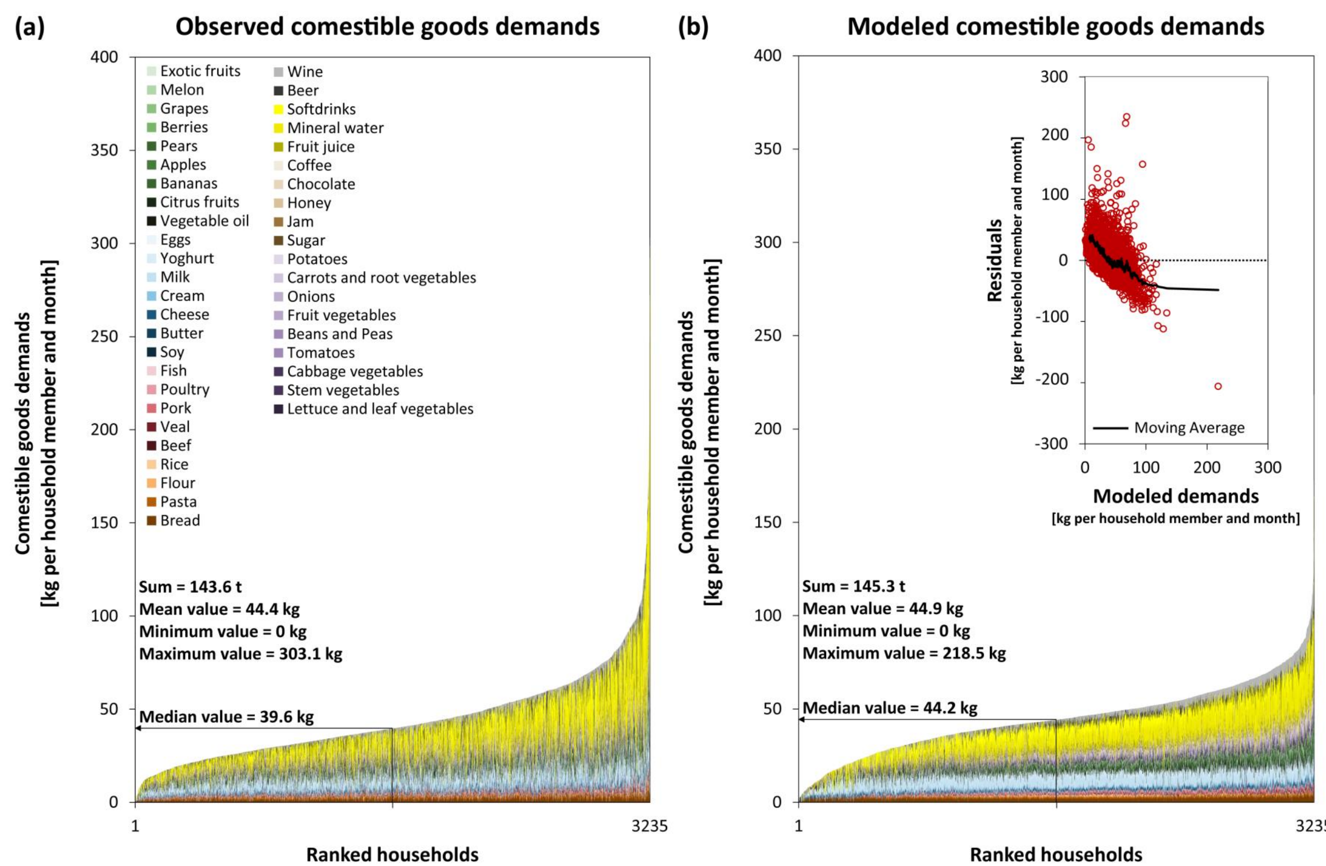

Figure 1. Observed (left) and modeled (right) demands of comestible goods for data reported in the Swiss household budget survey for the year 2004. Household demands are normalized by household size and ranked from smallest to largest demands.

A visual comparison of the two graphs (Figure 1) as well as the comparison of the five statistical parameters sum, mean, median, minimum and maximum value suggest that the accuracy of the modeling of demands is high. The relative differences between the statistical parameters for the observed and the modeled values are 1.2% for the sum, 1.1% for the mean, 0% for the minimum value. Only for the median (12%) and the maximum value (28%) the relative differences were larger. The accuracy was further investigated with a residuals plot (in the upper right corner of Figure 1b). It shows that the modeled values are slightly overestimated for low observed demands and more strongly underestimated for high observed demands. However, this is not a severe inaccuracy as observed high demands might also be a result of occurring events during the reporting period of that specific household. Events like birthday parties or anniversary celebrations might have led to extraordinary large food purchases.

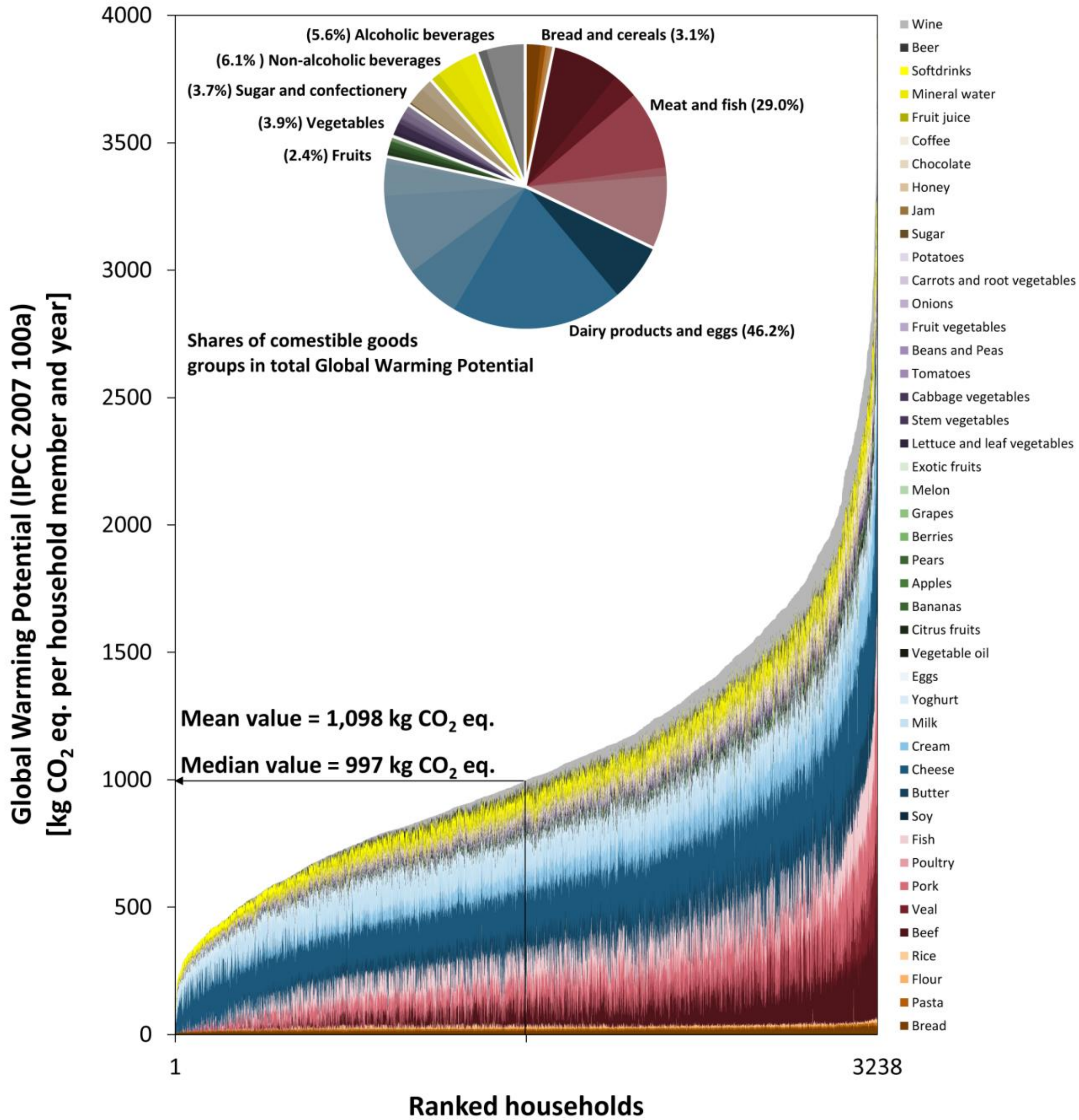

Figure 2. Global warming potential induced by in-home household consumption of 44 different food product and beverage groups in the case study municipality. The results are normalized per household member.

As we were able to reproduce the observed comestible goods demands with our two-tiered regression approach, we applied the method to a case study region. The results for the global warming potentials from at-home household consumption in Wattwil are shown in Figure 2. Greenhouse gas emissions are normalized by the number of people living in the households to adjust for different household sizes. Households are ranked according to the aggregated impact results. Results vary from 84 kg CO_2 eq. to 4.97 t CO_2-eq., with a median value of 997 kg CO_2-eq. and a mean of 1,098 kg CO_2-eq per year. The pie chart shows the shares of products in the total GHG emissions of the municipality. The consumption of meat & fish and dairy products are responsible for three quarters of total life cycle GHG emissions from at-home food products and beverage consumption in Wattwil.

Figure 2 shows that there is variability in the food carbon footprints. Some households consume almost no comestible goods at home, whereas others have significantly higher consumption than the rest. However, this variability is not so great compared to the variability of household environmental impacts in other consumption areas (Saner et al., 1013).

4. Conclusion and outlook

The food products and beverages demands of 3,238 individual households of a Swiss municipality were environmentally assessed. We found that the carbon footprints per household member and year vary from 84 kg CO_2 eq. to 5 t CO_2 eq. with a mean value of 1 t CO_2 eq. This variability is significantly smaller than the footprint variability for the consumption areas of housing and mobility, where 25% of the people are responsible for 50% of the environmental impacts (Saner et al., 1013). Differences between high- and low-impact households can be primarily explained by differences meat and dairy consumption, with vegetarian households in the lowest impact percentile, while the consumption of beverages and plant-based food was not decisive for the overall impact.

The presented study concentrated on at-home demand and supply of food products and beverages. The main reason was that for at-home consumption physical demand data is available from the household budget survey. However, we saw that only around 60% of food products and beverages are consumed at home. The rest is consumed in restaurants, cantinas, and takeaways, etc. Thus, future studies in this area should also focus on the modeling of out-of-home dining.

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Environmental assessment of urban horticulture structures: Implementing Rooftop Greenhouses in Mediterranean cities

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ABSTRACT

The Rooftop Greenhouse (RTG) of the Rooftop Greenhouse Lab (ICTA-UAB) is analyzed from an environmental perspective as a new form of urban agriculture. The global warming potential of an RTG structure was 2.5 kg of CO_2 eq., while the cumulative energy demand was of 46.4 MJ, considering a functional unit of 1m^2 and 1 year. When comparing the RTG structure with a multi-tunnel greenhouse, these values resulted in 80% and 53% higher, respectively. 1 kg of tomato produced in an RTG had a GWP of between 178 and 297 g of CO_2 eq. and a CED of between 2.9 and 4.8 MJ, depending on the crop yield. When compared with the horticultural production in a multi-tunnel greenhouse, 1 kg of tomato can be 33% less impacting or 25% more impacting for the GWP and 31% less impacting or 26% more impacting for the CED.

Keywords: rooftop farming, greenhouse technology, urban agriculture, smart cities

1. Introduction

Urban agriculture (UA) is spreading over the urban areas of developed countries in response to a growing awareness of the environmental impact of food systems (Howe and Wheeler 1999; Cohen et al. 2012; Mok et al. 2013). UA types are numerous and vary in placement, property and aim, such as community gardens for social inclusion, private backyard gardens for self-supply and public-property spaces for individual small gardens. Nowadays, UA is also colonizing buildings through building-based UA forms. These forms of UA have been defined by multiple authors as Vertical Farming (Despommier 2010), Skyfarming (Germer et al. 2011), Building-Integrated agriculture (Caplow 2009) or Zero-Acreage Farming (Specht et al. 2014).

Within the multiple forms of building-based UA, rooftop farming is the most common since rooftops are currently unused spaces that can be occupied and revalorized. Rooftop Greenhouses (RTGs) are greenhouses built on the rooftop of buildings devoted to, mostly, horticulture production (Cerón-Palma et al. 2012). Up to now, several companies in North America have built RTGs for their local production businesses. Gotham Greens (Brooklyn, New York) or Lufa Farms (Montreal) sell different kind of vegetables that have been produced in RTGs of 1400 m^2 and 2900 m^2, respectively, by offering, thus, km.0-products that avoid food-miles.

1.1. Current research on rooftop greenhouses

Only few studies have focused on rooftop greenhouses as urban horticulture systems. Cerón-Palma et al. (2012) identified the barriers to and opportunities of implementing RTGs in the Mediterranean region, by performing discussion groups with experts on architecture, agronomy and urban sustainability. Specht et al. (2014) did a literature review of urban horticulture in and on buildings, including RTGs, to determine the potentialities and limitations of these systems. Both studies found opportunities in the three pillars of sustainability: environment (e.g., reducing food-miles and transport emissions), society (e.g., improving community food security) and economy (e.g., revaluation of unproductive spaces). Notwithstanding the large potential benefits, barriers were also noted. Particularly, the studies highlighted social (e.g., lack of acceptance) and economic limitations (e.g., investment costs).

Sanyé-Mengual et al. (2013) quantified the potential environmental benefits of the local food production in terms of the avoided distribution of products from RTGs in Barcelona. A kilogram of tomato produced in a RTG in the city of Barcelona could substitute 1 kg of tomato from Almeria (1000 km), where 60$ of the tomatoes consumed in Barcelona are produced. The local production could avoid 441 g of CO_2 eq. and 12 MJ of energy per kg due to the optimization of the packaging use, and the reduction in the transport requirements and in the product losses. Moreover, RTGs can integrate their flows (i.e., energy, water, materials, gases) in the metabolism of the building. Cerón-Palma (2012) has worked on the quantification of the environmental benefits of the interconnection of the energy flow.

However, the implementation of greenhouses in cities must accomplish legal specifications of the urban context that imply an intensification of the materials consumption for the structure. Greenhouse structures in urban areas may face strict reinforcement, stability and security requirements. As a result, the greenhouse structure of RTGs may have larger environmental burdens than current horticulture production technologies, in contrast to the already quantified potential environmental savings of these systems in the literature (e.g., transport savings).

1.2. The Rooftop Greenhouse Lab (RTG-Lab)

The Rooftop Greenhouse Lab (RTG-Lab) is a RTG implemented for horticulture production in Bellaterra (Barcelona, Spain). The RTG-Lab is placed on the new building that houses the Institute of Environmental Science and Technology (ICTA) in the Universitat Autònoma de Barcelona (UAB). The RTG-Lab consists of two greenhouses of around 125m^2 each that are integrated in the rooftop of the building and aim to produce vegetables by means of soil-less techniques (i.e., substrate) (Figure 1).

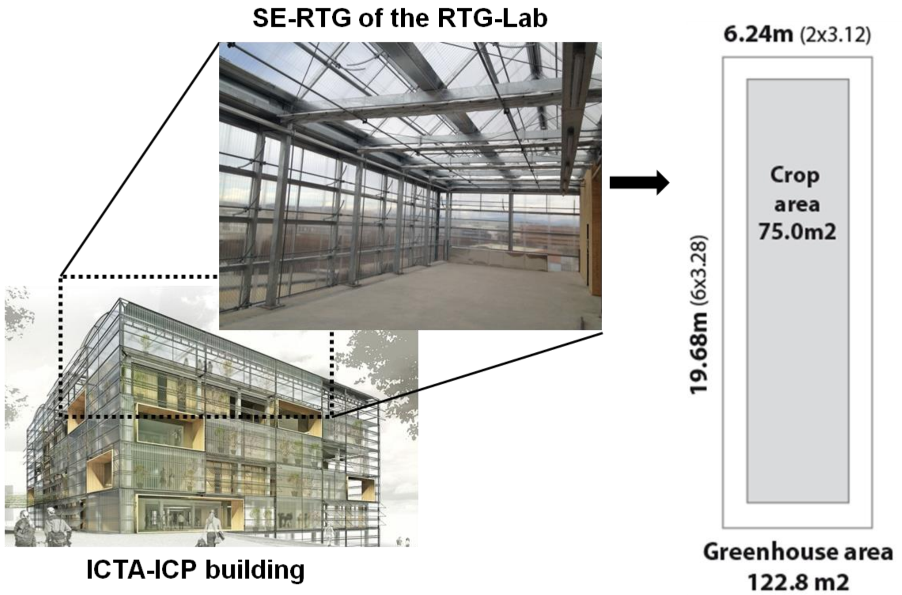

Figure 1. Situation of the RTG-Lab in the ICTA-ICP building, image of the South-East Rooftop Greenhouse of the RTG-Lab and RTG dimensions.

Research in the RTG-Lab will focus on three main issues: to proof the feasibility of crop production through RTGs in Mediterranean urban areas, to demonstrate and quantify the potential exchange of flows between the RTG and the building (i.e., energy, water, CO_2), and to quantify the environmental balance (both positive and negative impacts) of RTGs and their products. The RTG-Lab will start operating the late summer 2014 as well as the related projects. However, since the construction phase has already finished early environmental research regarding the RTG structure can be performed.

1.3. Objectives

The aim of the research is to perform an environmental assessment of the structure of a real RTG project in order to quantify the impacts of this new urban horticulture structure. To do that, the RTG of the RTG-Lab is analyzed through an attributional LCA that follows a cradle-to-grave approach. The assessment focuses on the structure itself (i.e., impact of the greenhouse life cycle) and on the agricultural production of the system (i.e., impact of the tomato production). Furthermore, the results are compared to a common industrial greenhouse structure of the study area, as a reference.

2. Methods

2.1. Goal and scope

This section presents the methodology of the study which follows the goal and scope definition of the ISO 14044 (ISO 2006).

2.1.1. Goal of the study

The aims of this LCA are to determine the environmental burdens of a Rooftop Greenhouse (RTG) as a new form of urban agriculture production structure and to compare it to the multi-tunnel greenhouse, which is one of the most used greenhouse structures in the industrial horticulture of the South Mediterranean region. The assessment also distinguishes between the greenhouse structure itself and the tomato production, where the greenhouse is an input.

2.1.2. Description of the system

The study focuses on the Rooftop Greenhouse of the RTG-Lab, as an urban agriculture system, although an industrial production system is used as a reference for comparative purposes. Moreover, this study considers the RTG of the RTG-Lab as an isolated RTG that do not exchange any flow with the building. For both systems, the system boundaries defined in the assessment are shown in Figure 2. The analysis of the greenhouse structures done from a cradle-to-grave perspective, where stages embraced are from materials extraction to the end of life of the structure. Secondly, attention is paid to tomato production and, thus, greenhouse structures become part of the life cycle of the agriculture production process. In this case, system boundaries are cradle-to-farm gate, which encompasses from the needed structure and equipment to the waste management. Tomato was chosen as horticulture product due to its importance in the vegetable market of the study area, where represents the second most sold product in MercaBarna (the food distribution center of Barcelona).

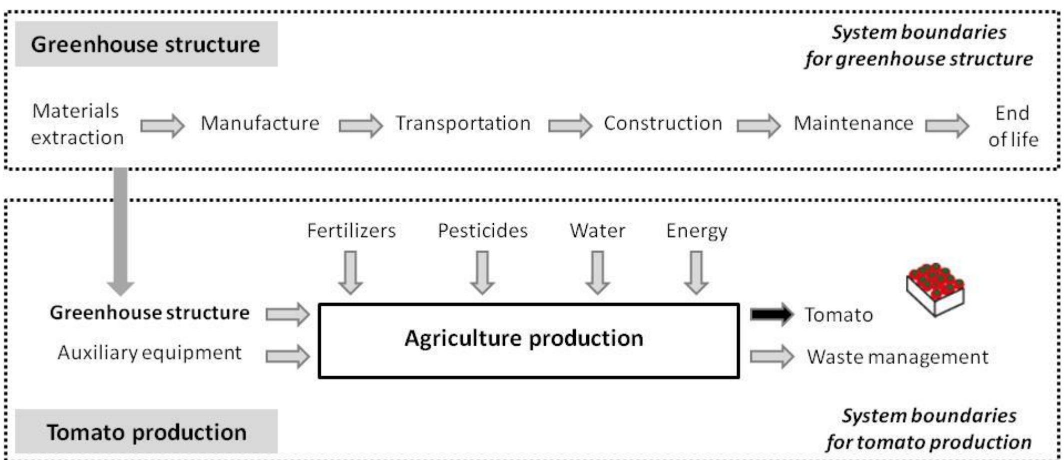

Figure 2. System boundaries of the assessment of the greenhouse structures and the tomato production.

The main differences between the RTG and the industrial system involve the greenhouse structure. The design and construction of the RTG-Lab had to overcome some legal barriers that imply a modification of the greenhouse structure: the Spanish Technical Code of Edification (CTE) (RD 314/2006 (BOE 2006)) and to fire safety laws (RD 2267/2004 (BOE 2004), Law 3/2010 (BOE 2010)). As a result, the RTG structure resulted heavier than an industrial greenhouse due to more intensive use of resources (e.g., steel). Moreover, Mediterranean greenhouses are light structures and commonly use LDPE covers, in contrast to glasshouses from colder areas (e.g., The Netherlands). However, the cover of the RTG-Lab was finally made of polycarbonate to be heavier but more resistant (particularly, offering a higher wind resistance). Finally, the construction requirements of an RTG are more intensive than for a soil-based greenhouse because materials must be raised to the rooftop.

2.1.3. Functional unit

The functional unit varies during the research as follows. First, when analyzing the greenhouse structure, the functional unit corresponds to 1m^2 of greenhouse structure for a timeframe of 1 year. Second, the functional unit of the analysis of the tomato production is 1 kg of tomato produced in a timeframe of one crop period, which is of 11 months in an RTG in Barcelona and 9 months in a multi-tunnel greenhouse situated in Almeria (Montero et al. 2011), due to the different climatic conditions. Although functional units correspond to 1 year, both assessments reflect the different lifespan of the greenhouse structures by including the maintenance stage. While the lifespan of the RTG is of 50 years according to project data and building elements, the lifespan of a multi-tunnel is of 15 years, according to law specifications (CEN 2001).

2.1.4. LCIA: method and impact categories

Two impact categories are included in the analysis. On one hand, the global warming potential (GWP, kg of CO_2 eq.) is calculated through the IPCC method (IPCC 2007) as an environmental indicator. On the other hand, the cumulative energy demand (CED, MJ) (Hischier et al. 2010) is accounted as an energy flow indicator. The SimaPro 7.3.3 program (PRé Consultants 2011) is used for the Life Cycle Impact Analysis (LCIA), which follows the classification and characterization phases defined as mandatory by the ISO 14044 regulation (ISO 2006).

2.2. Life cycle inventory

The life cycle inventory for the assessment of the rooftop greenhouse and the data collection process are detailed in the following section.

2.2.1. The Rooftop Greenhouse (RTG) system

(a) The greenhouse structure

The life cycle of the greenhouse structure consists of 6 stages, as illustrated by Figure 2: materials extraction, processing, transportation, construction, maintenance, and end of life. Data from the architectural project was used to quantify the amount of materials and their processing requirements. The transportation of materials was calculated according to the production site of each one, as reported in Table 1. The construction stage comprises the energy consumption of the machinery used to raise the materials to the rooftop and to build the greenhouse. Machinery consumed electricity from the grid and total consumption was calculated according to technical specifications. The maintenance of the structure was calculated based on the lifespan of the different materials, which according to producers' data were: 50 years for the steel and the concrete, 10 years for polycarbonate, 5 years for the climate screen, and 3 years for LDPE. In the maintenance stage, the amount of each material to complete the expected lifespan of the RTG (50 years) was considered. The end of life of the greenhouse structure includes both the transportation and the waste management. The structure is expected to be 100% recycled and recycling plants are 30 km far from the site.

Table 2 shows the life cycle inventory data of the Rooftop Greenhouse of the RTG-Lab for the entire greenhouse (122.8 m^2) and a lifespan of 50 years, and by functional unit (1m^2 and 1 year). The ecoinvent database v2.2 (Swiss Center for Life Cycle Inventories 2010) was used to complete background data of the LCI. The electricity mix for 2013 of Spain (REE 2013), of The United Kingdom (DECC 2014) and of The Netherlands (CBS 2013a; CBS 2013b) were used in the assessment of the materials processing.

Table 1. Origin, distance, and mode of transportation, by material of the RTG structure.

Material	**Origin**	**Distance (km)**	**Mode of transportation**
Steel	Martorell, Spain	77	Lorry 16-32t, EURO5
Polycarbonate	Doncaster, UK	1008.44	Transoceanic freight ship
		991.7	Lorry 16-32t, EURO5
Polyethylene	Tarragona, Spain	101	Lorry 16-32t, EURO5
Concrete	Barcelona, Spain	40	Lorry 16-32t, EURO5
Climate screen	Hellevoetsluis, The Netherlands	1487	Lorry 16-32t, EURO5

Table 2. Life cycle inventory of the 122.8-m^2 Rooftop Greenhouse of the RTG-Lab for a lifespan of 50 years, and for the functional unit of 1 m^2 and 1 year, by life cycle stage.

Flow	Unit	Total (50 years)	Per m^2 and year	Source
Materials (incl. maintenance)				
Steel (85% recycled)	kg	6430,54	1,05	Project data
Concrete	kg	1299,6	0,21	Project data
LDPE	kg	640,5	0,10	Project data
Polycarbonate	kg	985,74	0,16	Project data
Polyester	kg	47,3	0,01	Project data
Aluminum	kg	47,3	0,01	Project data
Lorry 35-40t EURO5	tkm	1723,3	0,28	Calculation
Transoceanic freight ship	tkm	994,07	0,16	Calculation
Construction			0,00	
Machinery use	kWh	2,32	0,00	Calculation
End of life			0,00	
Lorry 35-40t EURO5	tkm	284	0,05	Calculation
Recycling process	kg	9452,27	1,54	Project data

(b) The tomato production

The potential tomato production in the RTG of the RTG-Lab is assessed from a cradle-to-farm gate approach. Therefore, further life cycle stages, such as distribution and retail are excluded. Moreover, the RTG of the RTG-Lab is considered as an isolated RTG that do not exchange any flow with the building. The life cycle of the tomato production includes the equipment, the agriculture inputs and the management of the outputs (i.e., waste) as shown in Figure 2. The equipment consists of both the greenhouse structure and the auxiliary equipment, which includes the elements used in the crop system (i.e., perlite as substrate), in the irrigation (i.e., pipes, pumps, injectors, water distribution system, water tank) and in the inputs application (i.e., fertilizer tank). The application of fertilizers and pesticides includes their production as well as their emissions to air and water. Finally, the waste management accounts for the transportation requirements for the disposal of the outputs of the crop system, which are expected to be 100% recycled and recycling plants are 30 km far from the site.

Data for the auxiliary equipment was obtained from the EUPHOROS project (Montero et al. 2011), while fertilizers, pesticides and energy consumption was adapted from the same project by enlarging the crop period from 9 to 11 months. Water consumption was calculated through the “PrHo v2.0 for irrigation systems of greenhouse horticulture” of the Fundación Cajamar (González et al. 2008). No experimental data was available to determine the crop yield for producing tomato in a RTG, although the expected crop yield for a crop period of 11 months in the geographic context it is expected to be of 25 kg·m^{-2}, which combines a spring-summer and a summer-autumn crop cycles. However, since this value is still uncertain, two scenarios were assessed: an expected yield scenario (RTG), 25 kg·m^{-2}, and a low yield scenario (RTG$_L$), 15 kg·m^{-2}, to show potential constraints of crop production in RTGs, such as shadows from the structure or adjacent buildings.

2.2.2. Industrial horticulture: the multi-tunnel greenhouse

An industrial horticulture system is used as reference to compare with the RTG results. Horticulture production in Almeria is one of the most competitive horticulture production regions in Europe and, particularly, 60% of the tomatoes commercialized in MercaBarna (the food distribution center of Barcelona) are produced in Almeria. In this study, the multi-tunnel greenhouse structure and the tomato production in Almeria are used as reference for comparative purposes. LCI data for these systems were obtained from the EUPHOROS project, which assessed protected horticulture in Europe (Montero et al. 2011). According to the project, the crop yield considered for the industrial system is 16.5 kg·m^{-2}.

Table 3. Life cycle inventory of the tomato production in a RTG for 1 crop cycle (1 year) per area and per kg of product, by life cycle stage and crop yield scenario (RTG – 25 kg·m^{-2}; and RTG$_L$ – 15 kg·m^{-2}).

Flow	Unit	Per m^{2}and year	Per kg (RTG)	Per kg (RTG$_L$)	Source
Greenhouse structure					
Steel (85% recycled)	kg	1,05E+00	4,19E-02	6,98E-02	Project data
Concrete	kg	2,12E-01	8,47E-03	1,41E-02	Project data
LDPE	kg	1,04E-01	4,17E-03	6,95E-03	Project data
Polycarbonate	kg	1,61E-01	6,42E-03	1,07E-02	Project data
Polyester	kg	7,70E-03	3,08E-04	5,14E-04	Project data
Aluminum	kg	7,70E-03	3,08E-04	5,14E-04	Project data
Lorry 35-40t EURO5	tkm	2,81E-01	1,12E-02	1,87E-02	Calculation
Transoceanic freight ship	tkm	1,62E-01	6,48E-03	1,08E-02	Calculation
Machinery use	kWh	3,78E-04	1,51E-05	2,52E-05	Calculation
Lorry 35-40t EURO5	tkm	4,63E-02	1,85E-03	3,08E-03	Calculation
Recycling process	kg	1,54E+00	6,16E-02	1,03E-01	Project data
Auxiliary equipment					
LDPE	kg	2,30E-02	9,20E-04	1,53E-03	Montero et al. (2011)
Polystyrene	kg	2,60E-02	1,04E-03	1,73E-03	Montero et al. (2011)
HDPE	kg	9,40E-03	3,76E-04	6,27E-04	Montero et al. (2011)
PVC	kg	4,40E-03	1,76E-04	2,93E-04	Montero et al. (2011)
Steel (100% recycled)	kg	5,00E-04	2,00E-05	3,33E-05	Montero et al. (2011)
Expanded perlite	kg	6,20E-01	2,48E-02	4,13E-02	Montero et al. (2011)
Van, <3.5t	tkm	2,00E-04	8,00E-06	1,33E-05	Montero et al. (2011)
Inputs consumption					
Water	m^3	7,97E-01	3,19E-02	5,31E-02	Calculated
Electricity	kWh	1,08E+00	4,30E-02	7,17E-02	Adap. Montero et al. (2011)
Fertilizer (N)	g	9,76E+02	3,90E+01	6,51E+01	Adap. Montero et al. (2011)
Fertilizer (P_2O_5)	g	6,18E+01	2,47E+00	4,12E+00	Adap. Montero et al. (2011)
Fertilizer (K_2O)	g	1,91E+01	7,64E-01	1,27E+00	Adap. Montero et al. (2011)
Pesticides	g	4,00E+00	1,60E-01	2,67E-01	Adap. Montero et al. (2011)
Waste management					
Transport, van <3.5t	tkm	1,32E-01	5,28E-03	8,80E-03	Calculated

3. Results and discussion

3.1. Environmental assessment of the RTG structure

The global warming potential of an RTG structure is of 2.50 kg of CO_2 eq., considering the functional unit of 1m^2 for a timeframe of 1 year. The impact is mostly associated to the materials stage (including materials extraction, materials processing, transportation, and maintenance requirements) which contributes up to 99% on the GWP (Figure 3). The energy consumption during the construction phase and the transportation to the waste management site had, thus, a little impact on the entire life cycle of the structure. Consequently, the implementation of greenhouses on buildings has no large differences with industrial greenhouses on rural areas regarding their construction requirements.

Among materials, polycarbonate is the most contributor to GWP (54.7%), due to three factors: (a) the extraction and processing requirements (i.e., oil-based material); (b) the transportation requirements; and (c) the maintenance requirements that increase the polycarbonate consumption during the life cycle, even though other materials (i.e., steel) have a large presence in the structure in weight terms. For instance, steel contributes only to 29% of the GWP although representing the 68.0% of the total amount of materials. Polyethylene and the climate screen contribute to 10.4% and 5.2% of the GWP, respectively. Particularly, the transportation of the climate screen is the most contributing stage of this material. Finally, the anchor is the least contributing material to the GWP.

The cumulative energy demand of the system is of 46.4 MJ per m^2 and year. The distribution of the impact follows the same pattern as for the GWP regarding the life cycle stages (Figure 3). Among the materials, polycarbonate and steel are the most contributors, although the impact of polyethylene represents the 20% of the CED, because of the energy demand during the extraction and processing phases of this material.

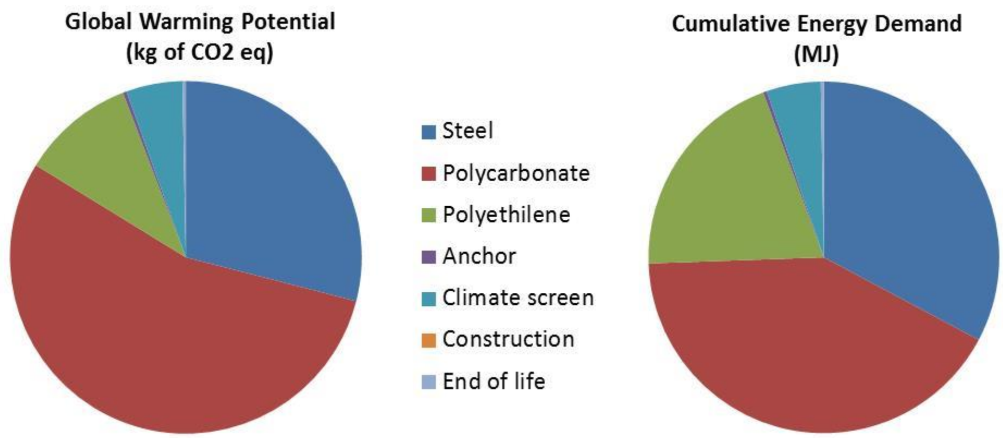

Figure 3. Distribution of the global warming potential impact and the cumulative energy demand of the RTG structure among the materials, construction and end of life stages.

In comparison to the industrial system, the global warming potential of an RTG structure resulted 81% more impacting than for a multi-tunnel greenhouse, while the cumulative energy demand was 53% higher (Figure 4). Main differences among the structures are of resources consumption for the materials. The RTG must accomplish more restrictive laws due to their situation in urban areas (e.g., security and resistance of building structures) and in height (e.g., wind resistance). For instance, the steel requirements per m^2 are 5 times higher in an RTG than in a multitunnel. Beyond law requirements, the analyzed RTG is an experimental system with an area of 122.8 m^2, while the industrial system operates in a 19440 m^2 commercial greenhouse. Therefore, certain "(in)efficiency of scale" produces a bias on the results for the RTG scenario and further case studies should be assessed in future research.

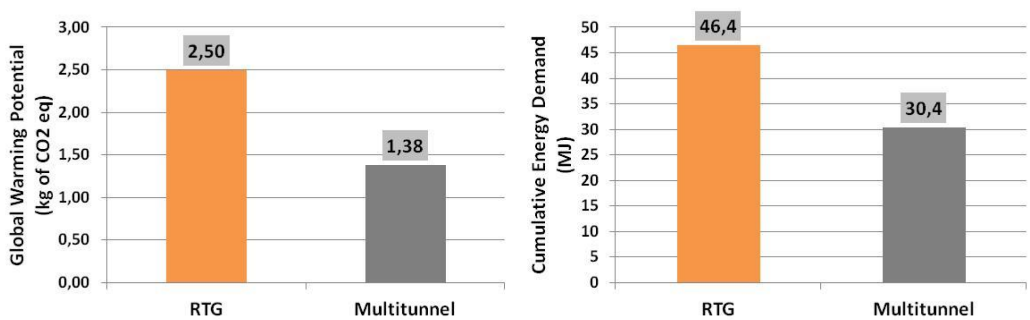

Figure 4. Comparative environmental assessment between the RTG and the multi-tunnel system, for a functional unit of $1m^2$ for a timeframe of 1 year.

3.2. Environmental assessment of the tomato production in RTGs

The production of 1 kg of tomato in an RTG has a global warming potential of between 178 and 297 g of CO_2 eq., depending on the crop yield. The RTG structure is the most contributor to the GWP (56.2%), a fact that is commonplace in horticultural systems (Torrellas et al. 2012), because of the large amount of materials consumed (e.g., steel and polycarbonate). Fertilizers and auxiliary equipment contribute to 22.8% and 17.1% of the GWP, respectively, since there is little energy and inputs consumption rather than fertilizers and water in an unheated greenhouse (Torrellas et al. 2012). Pesticides and waste management are the least contributors to the GWP value (<4%). The cumulative energy demand of 1 kg of tomato produced in an RTG is of between 2.9 and 4.8 MJ, depending on the crop yield, and the distribution of the impact among the life cycle stages follows a similar pattern as for the GWP indicator (Figure 5).

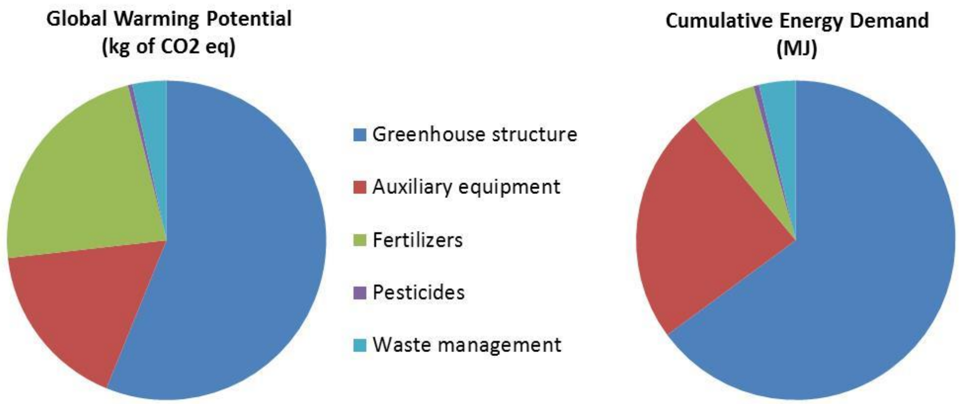

Figure 5. Distribution of the global warming potential impact and the cumulative energy demand of the tomato production in an RTG among the different life stages.

When comparing the tomato production in an RTG to industrial systems, the crop yield value becomes the most important variable. 1 kg of tomato produced in an RTG that has an expected yield of 25 kg·m^{-2} has a 33% lower GWP than produced in a multi-tunnel. However, assuming a crop yield of 15 kg·m^{-2} for an RTG can change drastically the results obtaining a GWP 25% higher per kg of tomato. The same results are obtained in the CED assessment, where the RTG scenario has a 31% lower value than the multi-tunnel, while the RTG_L scenario was 26% higher (Figure 6). Therefore, the assessment of the horticultural activity is sensitive to the crop yield considered, and there is a need to obtain experimental data from horticultural crops in RTGs in the Mediterranean area to reduce uncertainty in further studies.

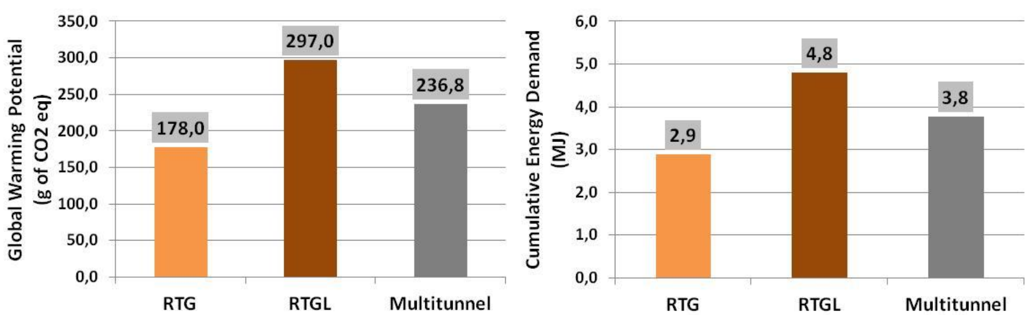

Figure 6. Comparative environmental assessment between the tomato production in a RTG with the expected crop yield – light orange; a RTG with a low crop yield – dark orange; and in a multi-tunnel system, for a functional unit of 1kg of produced tomato.

4. Conclusion

The RTG of the RTG-Lab (Bellaterra, Spain) was assessed to quantify the environmental burdens of a real case study of rooftop greenhouses to analyze new urban horticulture structures that are spreading over the developed world. The global warming potential of an RTG structure resulted of 2.5 kg of CO_2 eq. per m^2, where most of the impact came from the materials. Polycarbonate and steel were the most contributing materials to the total GWP. The construction and end of life stages of the life cycle of the structure were negligible to the final GWP value. The cumulative energy demand of the system was of 46.4 MJ per m^2, and the distribution of the impact followed the same pattern as for the GWP regarding the life cycle stages. When comparing the RTG structure to an industrial greenhouse structure, the impact was 80% higher for the GWP and 53% higher for the CED. This is caused, mainly, by the larger amount of materials used in an RTG structure due to legal requirements, which are more strict in buildings of urban areas than in rural areas (e.g., security).

The RTG system was also assessed as horticultural production system. The production of 1 kg of tomato in an RTG had a global warming potential of between 178 and 297 g of CO_2 eq., depending on the crop yield which is expected to be between 15 and 25 kg·m^{-2} (although no experimental data is still available). The cumulative energy demand of 1 kg of tomato produced in an RTG was of between 2.9 and 4.8 MJ. For both indicators, the greenhouse structure, the fertilizers and the auxiliary equipment were the most contributing elements. Compared to an industrial system, results were sensitive to the crop yield considered in the RTG. As a result, the GWP was 33% lower for 1kg of tomato produced in an RTG (25 kg·m^{-2}) than in a multi-tunnel system, although when produced in an RTG with a low yield the GWP of 1 kg of tomato resulted 25% higher. For the CED indicator, the RTG scenario obtained positive results (31% lower) when compared with the horticultural production in a multi-tunnel greenhouse, in contrast to the RTG_L scenario (26% higher).

Rooftop greenhouses (RTGs) had a larger environmental impact than industrial greenhouses. The main cause of the results was the law requirements in the urban context, which lead into an increase in the materials use in order to ensure the stability and resistance of RTGs. However, further case studies should be assessed in future research since the RTG-Lab has a largely smaller area than industrial systems. The environmental impact of horticultural production in RTGs is sensitive to the crop yield and, thus, showed benefits or negative impacts when compared with the production in a multi-tunnel greenhouse. Experimental data from horticultural crops in RTGs in the Mediterranean area are needed to reduce the uncertainty in future studies. Moreover, the integration of the RTG flows with the metabolism of the building can benefit the horticultural production by reducing the impact of water consumption (e.g., rainwater harvesting in building's roof) or increasing the crop yield (e.g., energy and CO_2 exchange). Nevertheless, further research and experimental data may demonstrate the potentialities related to integrated RTGs and their environmental balance.

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Moving from the ENVIFOOD Protocol to harmonized Product Category Rules and reference data: current and future challenges of the European Food Sustainable Consumption and Production Round Table

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ABSTRACT

The ENVIFOOD Protocol is a food and drink-specific guidance document created by the European Food Sustainable Consumption and Production Roundtable, a multi stakeholder initiative co-chaired by the European Commission and business associations from the food and beverage supply chains. The Protocol has been published in November 2013 and shall be used as a complementary guidance to the Product Environmental Footprint (PEF) and Organizational Environmental Footprint (OEF) guides in the PEF/OEF pilot testing launched by the European Commission. This paper describes the process of creating the ENVIFOOD Protocol as a consensus guidance document from the European food supply chain partners, and describes the key outcomes of the two public testing periods organized in 2013. Finally, we explain how the ENVIFOOD protocol is expected to be used as part of the PEF/OEF pilot testing, and the role it may play outside the application of the PEF/OEF guides.

Keywords: sectorial guidance, PCR/PEFCR, simplified ecodesign, harmonized environmental communication, Europe

1. Introduction

Lack of consistency in the methodologies for assessing and communicating the environmental performance of food and drink products has the potential to confuse consumers and other stakeholders involved in relevant supply chains. It also poses an unnecessary burden on organizations requested to evaluate their product's environmental footprint on the basis of different guidance leading often to different results. In order to address this issue, business associations, other food supply chain partners and the European Commission (EC) have established the European Food Sustainable Consumption and Production Round Table (RT).

This article provides an overview of the process to arrive at the Protocol, describes the process and outcomes of the public consultation and pilot testing and illustrates the next steps for the RT on the development and adoption of Product Category Rules (PCRs) in line with the Protocol and the EC's Product and Organization Environmental Footprint (PEF-OEF) guides. It also provides recommendations on the development of streamlined tools and an adequate database to best support such assessment tools. Finally, it provides insights on the future applications of the Protocol, especially in relation with the EC's PEF. The Protocol and PCRs will allow the development of user-friendly and affordable tools for assessment and communication of the environmental performance of food and drink products in Europe and beyond.

2. Methods: the process to create the ENVIFOOD Protocol

2.1. Setup of the Food Roundtable

The RT is co-chaired by the EC and food supply chain partners on equal footing and supported by the UN Environment Programme (UNEP) and European Environment Agency. When applying a life cycle approach, the RT's unique structure based on transparency and dialogue facilitates an open, results-driven and evidence-based dialogue among all players along the food chain which leads to further harmonization. The RT has delivered on its objectives according to schedule: the publication of the ten "Guiding Principles on the voluntary provision of environmental information along the food chain" (European Food SCP Roundtable, 2010), the Reports on "Communicating environmental performance along the food chain" (European Food SCP Roundtable, 2011) and "Continuous Environmental Improvement" (European Food SCP Roundtable, 2012) and the ENVIFOOD Protocol (European Food SCP Roundtable, 2013).

2.2. Creation of the ENVIFOOD Protocol

Since 2009, RT members have been working together on a commonly-agreed and science-based framework for assessment and communication of the environmental performance of food and drink products in Europe. Based on the above mentioned "Guiding Principles", the RT reached agreement on key methodological aspects at scientific workshops in 2010 and 2011 (Peacok et al., 2011; De Camillis et al., 2012). An analysis of relevant data, methodologies and guidelines for assessing the environmental performance of food and drink was also conducted. The analysis led to a harmonized methodology for environmental assessment, the ENVIFOOD Protocol. The Protocol provides guidance to support environmental assessments of food and drink products conducted in the context of business-to-business and business-to-consumer communication and the identification of improvement options. A public consultation period has been organized between November 21st 2012 and March 31st 2013. The consultation was specifically targeted at stakeholders in the food production chain, but open to anyone interested. The feedback from the public consultation was managed through the Roundtable Secretariat and addressed by subject matter experts from the Working Group 1.

2.3. Use of the ENVIFOOD Protocol in the PEF/OEF testing

The European Commission launched in January 2014 a second call for volunteers to test the development process of PEF/OEF guides. This second call was dedicated to food, feed and drink products. This call also included a testing of the ENVIFOOD Protocol in the development of the PEFCRs. The call closed on March 28th, the selected pilots presented in May, and the testing period started in June 2014. In this testing, the ENVIFOOD Protocol shall be used as a complementary guidance to the PEF/OEF guides (European Commission 2013). The RT will support the PEF/OEF testing as decided in the mandate for Working Group 1 for 2014, mainly on two axes: recommendations on the use of databases, as well as coordination of PCR/PEFCR development.

A first workshop on database development has been organized on June 11th 2014 in Brussels, and key database providers for the food sector have been invited to present their initiatives to the selected pilot testers as well as to interested stakeholders from the RT.

The working group will coordinate the development of product-specific rules (PEFCR/PCR) through the PEF pilot by:

- facilitating coordination and consistency between the pilots, including through participation in PEF pilot consultations and organization of technical workshops
- providing technical support for the interpretation of the ENVIFOOD Protocol, in relation with the EF Technical Helpdesk

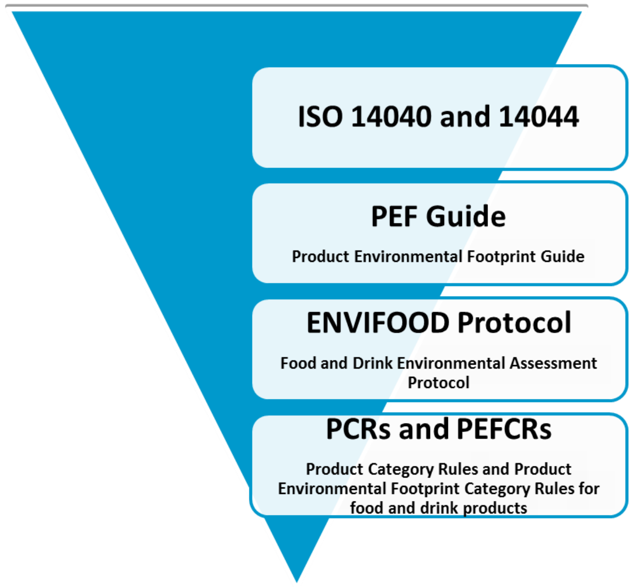

Figure 1. The intermediate position of the ENVIFOOD Protocol as a sectoral guidance in the context of the PEF/OEF pilot phase

As illustrated in Figure 1 above, the ENVIFOOD Protocol lies between the PEF Guide and the PCRs and PEFCRs. For example, assuming we are to assess on environmental performance of different coffee beverage products we would rely on the general guidance on LCA provided by the ISO norms 14040 and 14044 (e.g. the different phases of LCA). The methodology to follow would further be specified in the PEF Guide (e.g. which impact assessment model should be used). The ENVIFOOD Protocol then provides additional guidance specific to the food sector – in the example on coffee, this might be related to functional unit and the calculation of land use change associated to the development of coffee plantations. Finally, PCRs and PEFCRs would specify further details of how the assessment should be conducted at product level, including for instance on the consumer use phase (e.g. heating of water to drink the beverage). Based on all documents, calculation tools to assess environmental impacts of coffee beverages could be developed. Such tools would be sufficiently simple to use without the need of a deep understanding of all available guidance.

3. Results

3.1. Outcomes of the Public Consultation

A total of 11 stakeholders of different affiliations (industry, consulting, government agencies, and research institutes) have submitted feedback during the public consultation period. The feedback highlighted contradictions within the Protocol, misalignments with guidance provided by other institutions, further need for clarifications, as well as highlighting guidance with which certain stakeholders could not agree. Altogether, 144 comments have been received and have been analyzed by members of Working Group 1 of the RT to be included in the updated version of the ENVIFOOD Protocol. Many of the comments received could be used to improve the clarity of the guidance provided. However, a certain number of requests could not be followed because, among other reasons, a consensus on certain questions could not be achieved in the RT or because the points raised were considered more relevant for PCR than for the ENVIFOOD Protocol itself.

3.2. Outcomes of the Pilot Testing

A call for volunteers to evaluate the draft version of the ENVIFOOD Protocol has been made by the RT at the end of 2012. The pilot testing period lasted from March 27th 2013 to October 15th 2013 and actors of the food supply chain have been invited to test the ENVIFOOD Protocol in their organizations on case studies of new product developments or comparisons of existing products with product alternatives or competing products. The participants were free to choose the product to test. Also, the participants could choose whether or not to also communicate environmental performance based on the outcomes of the testing (the RT did not, at that time, recommend specific communication tools).

Overall, 18 organizations participated to the pilot testing. The participants included a wide range of manufacturers from the food and drink sector, research institutes, as well as trade associations. Written feedbacks were gathered from the 18 pilot projects and a workshop with pilot testers and RT members took place in February 2014 in order to reach consensus among the pilot testers on the modifications to include in the ENVIFOOD Protocol. The comments have been assessed by the Working Group 1 of the RT and classified into three categories:

1. Comments for immediate change, mainly concerning editorial comments or clarification needs. These comments have been incorporated into version 1.0 of the ENVIFOOD Protocol (published on Nov 20th 2013)
2. Medium term changes: some comments were requesting further guidance which could not be incorporated into the ENVIFOOD Protocol in the small amount of time available between the pilot testing and the publication of the Protocol. Therefore, these comments have been addressed in a separate guidance document which will be available in spring or early summer 2014 on the RT's website. In this document there will be a clear focus on Land Use Change (LUC) as the need for clarification on this topic was often mentioned (although the technical recommendations from the ENVIFOOD Protocol were not challenged)
3. Long term comments: some comments on fundamental questions could not be addressed, either because they would have required significant changes to the document, or because no consensus could be reached on them in the Working Group 1. They will be kept in a separate list and addressed during the next major revision of the ENVIFOOD Protocol.

3.3. Example: specific Results from the Testing of the Feed Sector

This paragraph describes the main outcomes of the pilot test of the ENVIFOOD Protocol undertaken by the EU feed industry, represented through the European Feed Manufacturers Federation (FEFAC), as an illustrative example of feedback from the pilot testing.

As an active member of the EU Food SCP, FEFAC contributed to the development of the ENVIFOOD Protocol. It was then a logical step to participate in the pilot test of ENVIFOOD Protocol in order to practically evaluate its relevance and applicability for the compound feed industry. This pilot test was undertaken by a consortium of feed associations and feed companies from the EU but also from outside EU.. Testing the recommendations of the ENVIFOOD by undertaking a concrete cradle to gate assessment for 21 feed compositions for feed for land animals as well as fish enabled to draw the following conclusions:

- It is currently necessary to be an experienced LCA practitioner in order to be able to deal with the requirements of the ENVIFOOD Protocol. Further work remains necessary so that these recommendations can be implemented by feed companies on a regular basis.
- The development of a comprehensive database, aligned with the requirements of the ENVIFOOD Protocol would be useful to facilitate the implementations of the requirements of the ENVIFOOD Protocol.
- For impact assessment, going through all the impact categories mentioned in the ENVIFOOD Protocol was not considered as something feasible. The selection criteria recommended in the ENVIFOOD Protocol were really useful to reduce the list to a manageable level.
- The recommendations regarding assessment of data quality were difficult to implement and considered as a way to try to quantify a subjective interpretation of data quality, which can often be just as well (or even better) discussed in a qualitative way. Moreover, there is no easy way to combine these types of quality indicators for a large range of data points (like in compound feed production) and end up in a meaningful cumulative assessment.
- The ENVIFOOD Protocol however provides added value when it comes to the environmental assessment of feed and constitutes a very relevant starting point for a feed PCR.

4. Discussion

The ENVIFOOD Protocol is also intended to be used outside the context of PEF/OEF. The main reason for this is that many actors in the food sector are interested in applying one single LCA methodology throughout their organization, which may be located in more than a country and even beyond the EU. Therefore, if the ENVIFOOD guidance is implemented as part of the PEF/OEF approach in these organizations, they would like to also use the same guidance outside the context of PEF/OEF. The RT therefore is in contact with organizations inside and outside Europe that work on sustainability in the food sector, and is interested in promoting the guidance developed in the RT also in other organizations.

5. Conclusion

The RT as a large European stakeholder initiative has successfully established scientifically solid and harmonized guidance on life cycle assessment for the food and drink sector. A number of guidance documents have been published (ENVIFOOD Protocol, but also related documents on communication, databases, PEFCR/PCR development, etc.) over the past years. While the ENVIFOOD Protocol and associated documents do not solve all challenges in assessing the environmental impacts in the food sector, the RT has created a platform for exchange of views between stakeholders, and has managed to establish a methodology that can further evolve as consensus forms and scientific methods improve.

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Does the environmental gain of switching to the healthy New Nordic Diet outweigh the increased consumer cost?

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ABSTRACT

The New Nordic Diet (NND) was designed by gastronomic, nutritional and environmental specialists to be a palatable, healthy and sustainable diet containing 30-40% less meat than the Average Danish Diet (ADD), ≥75% organics, and more locally grown wholegrain products, nuts, fruit and vegetables. In this study, the NND was based on economic modelling to represent a "realistic NND bought by Danish consumers". The objective was to investigate whether the ADD-to-NND diet-shift has environmental consequences that outweigh the increased consumer cost of the diet-shift. The diet-shift reduced the three most important environmental impacts by 16-22%, mainly caused by reduced meat content. The surcharge to consumers of the ADD-to-NND diet-shift was €216/capita/year. In monetary terms, the savings related to the environmental impact of the diet-shift were €151/capita/year. 70% of the increased consumer cost of the ADD-to-NND diet-shift was countered by the reduced socioeconomic advantage associated with the reduced environmental impact of the NND.

Keywords: ADD (Average Danish Diet), Environmental impact, LCA (Life Cycle Assessment), NND (Optimal well-being, development and health for Danish children through a healthy New Nordic Diet), Socioeconomic cost.

1. Introduction

On a global scale, agricultural production consumes large amounts of resources and releases large amounts of greenhouse gases (7.3–12.7 Gt CO_2-eq, or 14–24% of total global emissions; Vermeulen et al. 2012), air pollutants, nutrients, and pesticides. In 2011, Danish agriculture contributed with 0.01 Gt, or 17% of the total Danish greenhouse gas emission (Nielsen et al. 2010). Agricultural production alters soil structure and carbon storage in the soil, contributes to eutrophication, diminishes biodiversity, and causes unintended toxic effects on flora and fauna, including humans. Whereas the growing and production of feed, food and beverages have serious impacts on the environment, we all have to eat and drink. But what we choose to eat and drink greatly affects the environmental impact on ecosystems, human well-being and resource expenditure. Our choice of diet is our own, although it is often associated with ethnicity, social status, habit, age, and sex and is influenced by policy and economics (Steptoe and Pollard 2005).

Production of livestock and dairy products typically cause greater environmental impacts than the production of cereal, fruit, vegetables and legumes (Audsley et al. 2009, Tukker et al. 2011, Saxe et al. 2013, Saxe 2014). Reducing the content of animal produce, particularly meat, and increasing the content of grain products, fruit and vegetables in the typical Western diet would decrease the environmental impact of eating and drinking (Weidema et al. 2008).

This study is part of the Danish multidisciplinary OPUS project that develops, tests, and aims at disseminating a New Nordic Diet (NND). The NND was designed by gastronomic, nutritional and environmental specialists to be a palatable, healthy and sustainable diet of Nordic origin containing 35% less meat than the Average Danish Diet (ADD), more fish, wholegrain products, nuts, fruit, berries and vegetables, locally grown food in season, and more than 75% organic produce (Mithril et al. 2012, 2013, Poulsen et al. 2014). The impact of an ADD-to-NND diet-shift on climate change (Saxe et al. 2013) and on a wider range of environmental impacts (Saxe 2014) has already been investigated. The first of these studies was based on the OPUS dietary recommendations (Mithril et al. 2012), while the second was based on 180 OPUS recipes for the NND.

In the present study we apply an economic model to fulfill the above-mentioned NND dietary recommendations in the most incentive-compatible way, i.e. composing the most realistic NND bought by the population of Danish consumers.

The Objective of this study was to investigate the environmental consequences of an ADD-to-NND diet-shift – in “real life” – and if the socioeconomic value of these consequences outweigh the increased consumer cost of the diet-shift.

2. Methods

The ADD is the reported Danish consumption of foods and beverages in 2010 as represented in the consumer household survey published by Statistics Denmark (2013). The consumer survey displays the composition of the food and beverage budget on COICOP (United Nations, 2014) commodity categories, and these budget components are converted to physical quantities using consumer price data estimated on the basis of household purchase data from a commercial market survey company (GfK).

The ADD serves as the point of departure for determining the composition of the NND. In particular, the composition of the NDD is estimated by adjusting the items in the ADD by means of an economic simulation model developed for the purpose, which describes the consumers' preferences, such as preference-based substitutability between different food and beverage commodities. For example, if two commodities are close substitutes, an increase in the price of one of these products would induce a relatively strong shift in the consumption of these two commodities – the consumption of the commodity with increased price will decrease relatively strongly and the consumption of the other product will increase. In contrast, if a commodity is not very substitutable with other products, the consumption of this product will only be affected to a limited extent. Hence, sensitivity to price changes (measured by price elasticities) reflects commodity substitutability. Because substitutability between food and beverage commodities - both with regard to nutritional value and with regard to their appeal to consumers' preferences - varies significantly, the adjustments in the consumption patterns will also vary accordingly. Price elasticities (reflecting substitutability between food and beverage commodities) have been estimated econometrically on the basis of the above-mentioned commercial market survey data from GfK for five income classes of households.

Compared with the ADD, the NND involves a number of restrictions, including lower bounds for the intake of some products (categories of fruit, vegetables, seafood and whole-grains) and upper bounds for other elements (meat, sugar, saturated fat). These restrictions are implemented in the economic model by calculating the set of (implicit) prices that would be consistent with the restricted diet in utility-maximizing equilibrium. In addition to ensuring compliance with the requirements to the NND, the implicit prices also induce specific changes in the consumption of each individual commodity, as a consequence of the above considerations about substitutability. Hence, the consumed quantities are estimated by adjusting the ADD figures by means of implicit price changes and price elasticities, and these estimated quantities are multiplied by the original market prices in order to calculate the households' food budget in the NND.

The environmental impact of the ADD and the NND was compared based on 15 impact categories (carcinogenic and non-carcinogenic toxicity, respiratory inorganics, ionizing radiation, ozone layer depletion, aquatic and terrestrial ecotoxicity, nature occupation, global warming, acidification, aquatic and terrestrial eutrophication, respiratory organics, photochemical ozone effects on vegetation, and non-renewable energy; Weidema 2009), which were monetized to evaluate the overall effect in “shadow price” associated with an ADD-to-NND diet-shift. In this context, the term “shadow price” signifies the environmental cost of the diet-shift.

The comparison was based on consequential life cycle assessment (cLCA) using the Simapro 8 software, and the international Ecoinvent (Ecoinvent 2014) and the Danish LCA food databases (2004), and the Stepwise method to calculate the environmental impact and the external environmental cost (shadow price) (Weidema et al. 2008, Weidema 2009). When the LCA food and Ecoinvent databases lacked information, supplementary data that best fit the Danish production conditions were taken from the literature. All environmental impacts were for each food or beverage item calculated according to the ISO standard 14040 (2006). The consumer price was found in order to calculate the consumer cost of the diet-shift in order to relate this to the shadow price.

CLCAs seek to identify the environmental consequences of a decision or a proposed change in a system under study (oriented to the future), which means that market and economic implications of a decision may have to be taken into account (Earles and Halog 2011). The functional unit was 1 person year’s diet measured in kg manufactured food and beverage products. Waste is included in the calculations to the extent that the diets are made up of what is produced for diets, not what is consumed, i.e. the waste from the farm gate to the table is included. The scope of the study included the response of 15 environmental impact categories associated with all activities, energy, and resource consumption from soil to supermarket. However, for clarity, only data for the three most important impact categories (in terms of monetized impacts) is presented in the Figures 1 to 3.

The ADD was the reference against which the environmental impact of the NND was measured. In this study, the ADD was represented by 66 food and beverage products or categories supplied to the average Dane for private consumption. However, the products were pooled into 53 categories to fit the available cLCA data. For the graphic representation in Figures 1 to 3, all foods and beverages were further pooled to into 11 categories to ensure a lucid presentation.

In the present paper, we only study effects of the ADD-to-NND diet-shift based on the difference in the dietary composition, while in previous NND studies the commodity import distances and production strategy (conventional vs. organic) were taken into account (Saxe et al. 2013, Saxe 2014). At the present level of aggregation it neither makes sense to include the impact of import distances nor to include effects of the production strategy since these vary between individual products in each of the product categories assumed for this study. As an example the category "cheese" include imports from several different countries (import distances), where cheese of each origin can be of either conventional or organic origin, all sold to consumers in Denmark. In the present study, the focus is on the "real life" NND modeled to be bought by Danish consumers, its consumer price and shadow price.

Substituting animal produce with vegetables, legumes, whole grain products and fruit may reduce the intake of protein and some essential nutrients. In this study the ADD and the NND had similar energy and protein contents. This was obtained using the above-mentioned price elasticities for backward calculation of "implicit prices". These implicit prices represent the prices that would give the consumers the incentive to choose a diet with unchanged energy and protein contents, but with a dietary composition consistent with the NND specifications. But in addition to ensuring fulfillment of the NND-specifications, these implicit prices also determine the consumption of individual products within the commodity groups, which was the basis for the cLCA. Hence, only diets with similar energy, protein and nutrient content may be directly compared.

For effective and comprehensible presentation of the potential of the ADD-to-NND diet-shift in reducing the environmental impact of diets, the annual mileage driven in a Euro class 5 passenger car was used reference (Ecoinvent 2014).

We tested one more diet in this study – a so-called "SensWell" modification of the ADD diet (SW-ADD). SensWell is a research project that develops and tests new healthy and satisfying foods and drinks that though improved taste may substitute unhealthy foods and drinks in the daily diet. In the SW-ADD, soft drinks are replaced with a theoretical designer drink with a high *umami*. *Umami* is the 5th faculty of taste (besides sweetness, sourness, saltiness and bitterness), commonly found in its salt form as the food additive monosodium glutamate. For that reason, scientists consider *umami* to be distinct from saltiness. *Umami* is detected through specialized receptor cells present on the human and other animal tongues due to detection of the carboxylate anion of glutamate. *Umami* can be described as a pleasant "brothy" or "meaty" taste with a long lasting, mouthwatering and coating sensation over the tongue. 0.2 liter of the designer drink was assumed to be consumed by a third of the mature population every workday to replace soft drinks at "snack time". Our models show that it would not only replace a certain amount, i.e. about 15% of the total Danish consumption of soft drinks, but also other beverage and food, noticeably meat, e.g. beef and pork because of its high "*umami* content".

3. Results

The composition of the studied ADD and NND are given in Table 1. According to the NND dietary recommendations, tomatoes, cucumbers, coffee, tea, cocoa, wine, beer, and spirits of non-Nordic origin should not be part of the NND since they are not of Nordic origin. However, in the present version of the NND they are accepted at the level recommended in the Danish dietary guidelines (Astrup et al. 2005) based on the expectation that few people will do without these commodities in "real life". And as already stated, this paper aims at studying the Danish implementation of the NND in "real life". Table 1 shows that the modeled NND contains 39% less meat than the ADD, which are the OPUS NND recommendations of a 30-40% reduction.

The NND mass is 13.9% larger than that of the ADD mainly caused by a higher content of fruit and berries, vegetables and grain products. Fruit, berries and vegetables have higher water content than most other commodities and their mass is together with grain products genuinely larger in the NND (Figure 1).

Table 1. The mass and consumer price of each of the 53 food and beverage products or categories supplied to the Danish population in 2010 named the Average Danish Diet (ADD). The ADD is the reference to the modeled New Nordic Diet (NND)

Ingredients	ADD, Kg/ capita/y	ADD, €/ capita/y	NND, Kg/ capita/y	NND, €/ capita/y
Apples	8.6	22.0	13.7	39.1
Bananas	7.4	19.0	11.6	30.1
Beef, veal	13.8	118.3	9.4	80.6
Beer	57.1	74.6	74.3	101.4
Berry fruits	2.1	32.7	16.7	319.9
Bread, sand cakes, cookies, biscuits, pasta products	58.4	207.5	42.9	141.4
Butter, butter containing spreads	4.4	29.0	0.0	0.0
Cabbage	7.5	13.4	10.7	18.3
Canned fruits, fruit salads	1.4	3.8	1.4	3.7
Cheese	13.8	109.3	12.5	102.1
Chocolate (dark, not filled)	4.9	61.0	4.6	59.1
Citrus fruits	10.5	20.7	39.1	116.9
Coffee, tea, cocoa	9.4	63.2	8.0	73.8
Dried fruit, nuts	2.7	39.7	2.7	39.7
Dried vegetables	0.5	2.1	0.5	2.1
Eggs	9.4	31.0	9.6	31.6
Flour, grains	105.8	58.2	179.1	77.0
Fresh and frozen fish	1.6	31.9	2.1	40.5
Fruit juice	22.9	34.0	22.8	34.3
Ice cream	20.6	37.7	16.9	31.0
Jam, honey, candy, raw marzipan, other sugar products	14.9	101.6	12.3	97.7
Lamb	0.91	13.2	0.9	12.8
Lettuce, Chinese cabbage, parsley	5.4	24.9	5.7	27.5
Margarine, all kinds	5.7	11.5	2.4	4.9
Mineral water, incl. soft drinks	79.7	80.5	82.1	81.9
Other fresh meat	0.3	2.7	0.3	2.7
Other fruits	4.1	16.0	3.0	12.0
Other milk products	8.1	32.5	8.0	32.2
Peaches, plums, cherries, avocados	8.9	17.1	13.6	27.6
Pears	1.4	5.9	1.9	8.7
Pizza, spring rolls, other cakes	6.6	35.8	4.3	25.6
Pork	11.0	88.9	6.3	50.9
Pork fat	0.3	0.6	0.3	0.6
Potato products	2.1	15.2	1.7	12.4
Potatoes	32.3	28.2	49.0	44.2
Poultry	9.2	66.5	3.6	26.2
Processed and mixed vegetables	13.1	38.0	9.3	26.8
Processed fish, fish products)	8.3	50.1	10.2	60.7
Processed meat	3.5	23.3	3.5	23.2
Rice	7.9	10.4	8.0	10.5
Root crops, onions, mushrooms	24.6	43.8	54.2	77.2
Semi-skimmed, skimmed & buttermilk, infant formula	92.3	61.9	72.1	48.0
Shellfish (not canned)	1.1	10.8	1.5	15.3
Smoked and salted fish	0.6	12.4	0.6	13.4
Soup, sauce, bouillon, flavor products, yeast, preservatives	19.6	72.1	19.7	73.3
Soured whole milk, yoghurt	22.2	36.0	19.8	32.2
Sugar	3.5	8.9	0.7	1.9
Tomatoes, cucumbers, pepper bells, peas	18.1	62.1	19.7	69.6
Variety & cold meat, bacon, sausage	13.6	161.7	7.9	94.8
Vegetable juice	0.4	2.2	0.3	2.1
Vegetable oils	3.2	8.9	4.	11.5
Whole milk	11.3	10.3	8.1	7.3
Wine, port-, fruit- & dessert wine, champagne, spirits	44.5	206.3	44.5	206.6
Total	**841**	**2369**	**958**	**2585**
NND increase relative to the ADD	**100%**	**100%**	**13.9 %**	**9.1 %**

The NND reduced the environmental impact relative to the ADD measured by 12 of the 15 impact categories. The socioeconomically most important impacts (in terms of monetized impacts: respiratory inorganics, i.e. fine particles < 2.5 µm in aerodynamic diameter ($PM_{2.5}$), nature occupation and global warming) were all decreased by 16% to 22%, mainly caused by the reduced meat content in the NND (Figure 1). According to the LCA food database (2004) and the Ecoinvent database (2014), meat has a higher environmental impact per kg than most other commodities, and therefore dominates the environmental savings associated with the ADD-to-NND diet-shift.

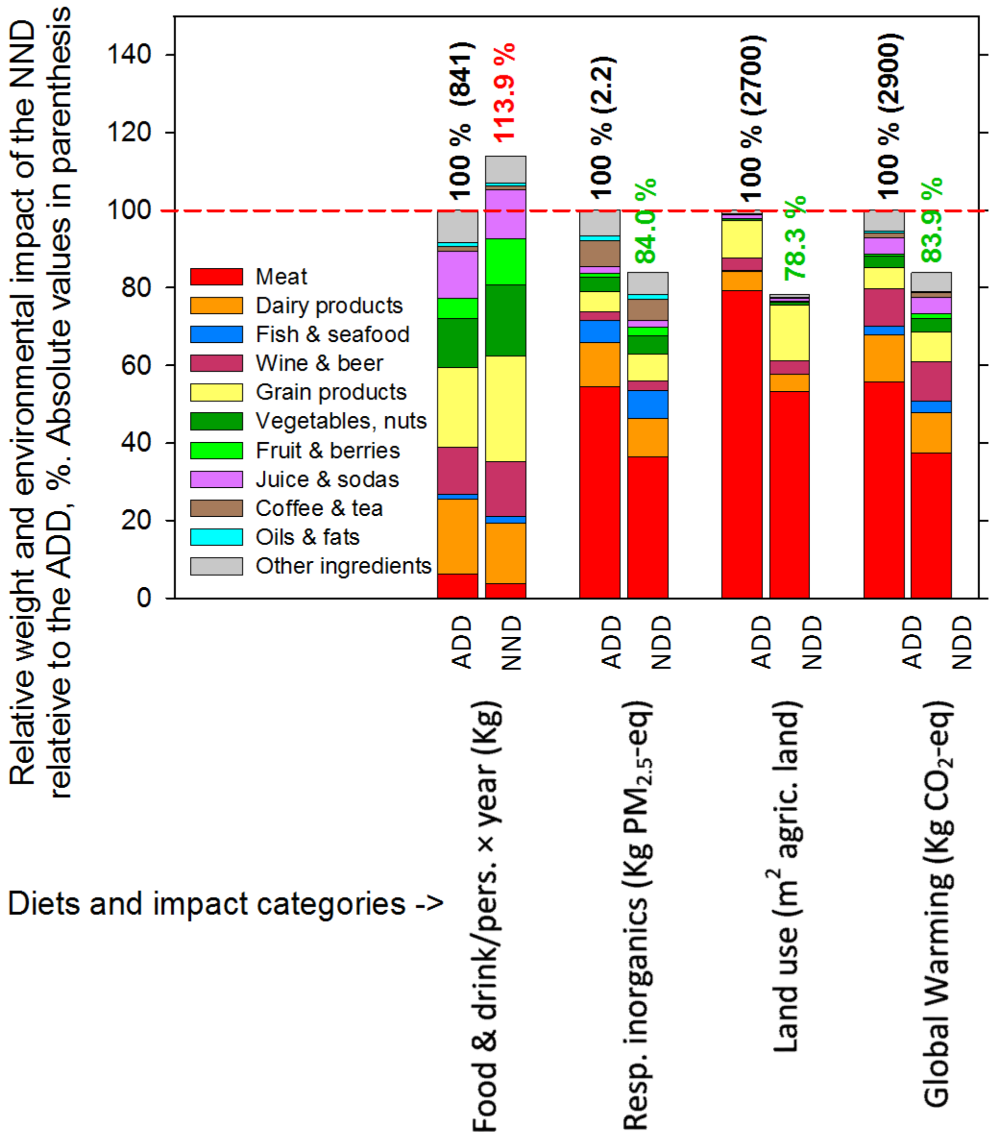

Figure 1. The quantities consumed and selected environmental impacts of the Average Danish Diet (ADD) and the modeled New Nordic Diet (NND) showing the environmental benefit of the ADD-to-NND diet-shift.

The absolute values in Figure 1 are given in parenthesis, e.g. 2900 kg CO_2-eq released per capita and year with the ADD, and 16.1% less for the NND (2440 kg CO_2-eq). It should be noted, that the land use change (LUC) was included in the global warming calculations as implemented by Saxe (2014). The inclusion of LUC (using values taken from Audsley et al. 2009) more than doubled the difference between the GWP of the ADD and the NND and nearly doubled the difference between the overall (monetized) environmental impacts of the two diets (Saxe 2014). The content of meat and dairy products in both diets dominate in each of the three environmental impact categories.

The environmental cost to society (shadow price) of the ADD was found to be €820 and that of the NND €669 (Figure 2). The ADD-to-NND diet-shift therefore potentially saves society for €151/capita/year in terms of improved environmental conditions. Figure 2 shows that the meat content of both diets dominate the shadow price most through its impact on nature occupation and global warming and least through the sum of remaining impact categories. The second most important impact on nature occupation is grain products. The ingredients in the "sum of remaining impact categories" that dominate the environmental cost (shadow price) are sweets, coffee and cocoa. Fish and seafood have its highest relative impact on respiratory inorganics ($PM_{2.5}$; Figure 1, 2). This impact is caused by the considerable diesel consumption of fishing boats and the ice for onboard storage of the catch.

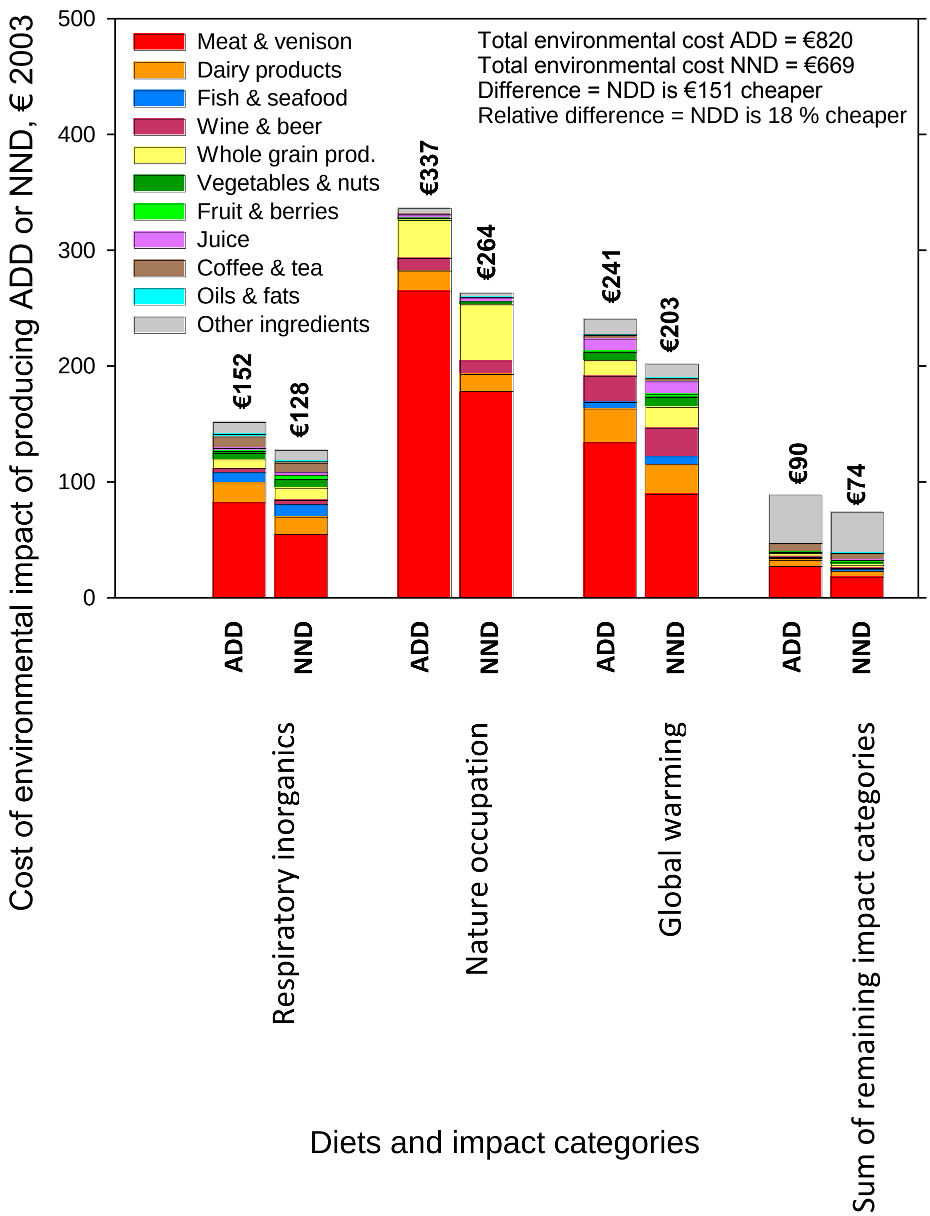

Figure 2. The potential environmental cost to society (shadow price) of the Average Danish Diet (ADD) and the modeled New Nordic Diet (NND).

Figure 3 compares the consumer price with the shadow price (potential environmental cost) of the two diets. The consumer price of the ADD was €2369/capita/year and of the NND €2585/capita/year when neither the price premium for organic production nor the savings by (mainly) having local produce in the NND was included (Figure 3). The increase in consumer price associated with the ADD-to-NND diet-shift was therefore €216 /capita/year, or a 12 % increase in consumer price, compared with ADD.

The potential savings (€151) reflected by the shadow price of the NND cover 70% of the increased consumer price (€216) for the NND. The environmental cost of driving a Euro class 5 car 1 mile was found to be 0.078 € per mile (Ecoinvent 2014). The savings caused by the diet-change (mainly caused by the lower meat content in the NND), equals the environmental impact of driving a EURO class 5 passenger car 1935 miles per year, i.e. a quarter of the average annual mileage for a Danish passenger car.

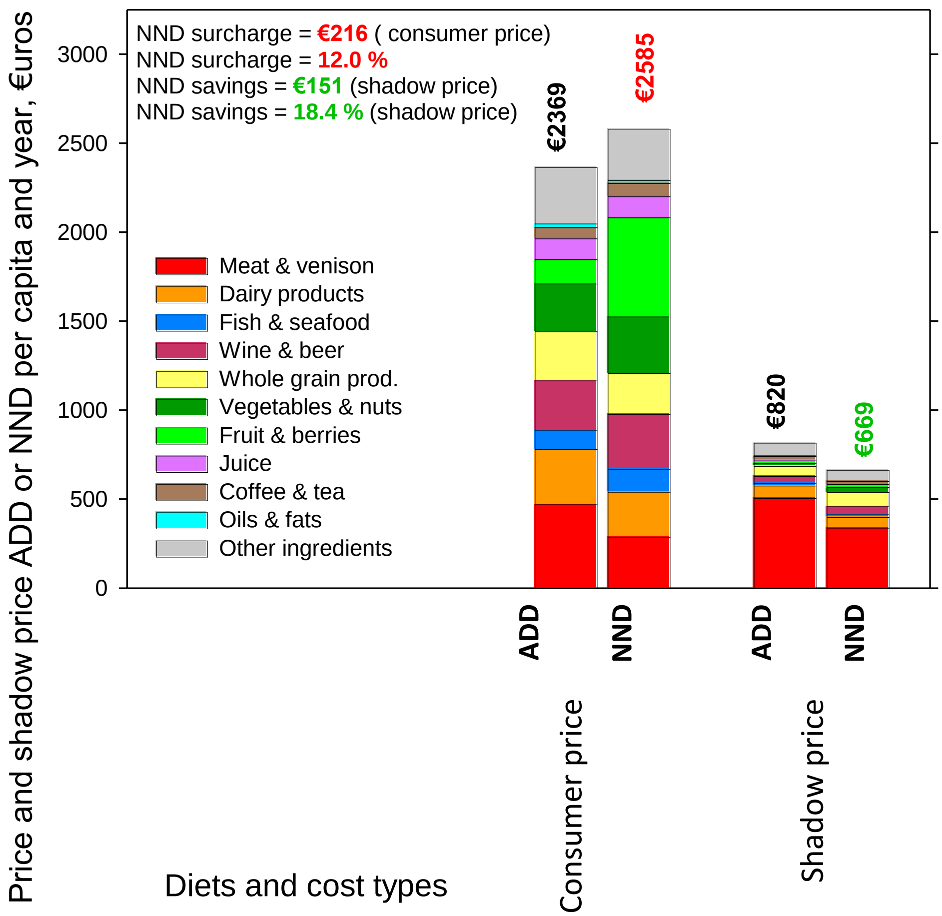

Figure 3. Consumer price of the modeled Average Danish Diet (ADD) and New Nordic Diet (NND) compared with the shadow price of the same diets.

Figure 3 also shows that the shadow price of the meat content in both diets exceeds the consumer price. For the ADD the shadow price exceeds the consumer price by €35.78 or 8%. For the NND the shadow price exceeds the consumer price by €51.46 or 18%. Reducing the meat content in either diet is consequently the most effective way to lower the impact of diets seen in a socioeconomic perspective.

The SW-ADD increased the kg intake relative to the ADD by 0.8%, while the main impact categories were decreased by 1.9-2.8% (data not shown). The consumer price for the SW-ADD increased by €96.54 or 4.1 %, over the ADD consumer price, while the shadow price decreased by €22.32 (data not shown), compensating for about a quarter of the consumer price increase.

4. Discussion

The prerequisite for trusting in the calculated reduced shadow price associated with the ADD-to-NND diet-shift is that (1) the Ecoinvent and the LCA food data are adequate for the purpose, and (2) that the Stepwise method calculates the environmental impacts correctly, and monetizes these impacts correctly. In this study, we assume both prerequisites to be fulfilled. It should however be mentioned that there are significant variations in findings regarding the economic value of some of the environmental benefits across studies, where some studies come out with lower shadow prices than those used in the present study.

The principles behind the OPUS NND have proved to be of great advantage to the environment, and to the potential socioeconomic savings associated with this healthy diet. The shadow price of the ADD-to-NND diet-shift was found to cover 70% of the increased consumer price of the diet-shift. In this study we have not included health advantages of the diet-shift, but since the NND was created to be a healthier diet than the ADD, it can be presumed that there will also be a socioeconomic advantage due to improved health when choosing the NND. All in all it may be cheaper to consume the NND than the ADD seen from a societal point of view – and this should be reflected in the price we pay for the ingredients of our diet. To have a direct consequence for the consumer prices, the Rio Declaration Principle 16 (1992, "the polluter pays principle") should be implemented for food consumption. Animal produce should be more expensive and vegetables and fruit cheaper. That would motivate more consumers to protect the environment and improve their health through their free choice of diet.

With the above in mind, it makes sense to consider further steps that make our diets even more environmentally friendly and socioeconomically beneficial. Saxe (2014) found that a vegetarian version of the NND could reduce the GWP by 67% when transport associated with imports of both the ADD and the NND was taken into account and by 59% when an 84% content of organics in the NND and the actual 8% content of organics in the ADD was taken into account. These reductions are more impressive than in the present study. One reason for this is that the ratio of meat types in the NND study by Saxe (2014) based on NND recipes was more advantageous. The recipes took into account that the production of beef, in particular (Cederberg et al. 2011), and pork is more harmful to the environment than is the production of grass-fed lamb, poultry, or fish. Relative to the distribution of meat types in the ADD, the NND in Saxe (2014) included only 30% beef and veal, 36% pork, and 73% chicken, but 680% grass-fed lamb and 820% venison. In the present study the meat content relative to the ADD was 68% beef and veal, 57% pork, 39% chicken and 98% lamb. Though the overall meat reduction in the present study was 39% vs. a 35% meat reduction in the study by Saxe (2014), the smaller reduction in beef and pork resulted in a smaller reduction in environmental impact in the present "real life" version of the NND. Another contributing factor to a lower effect of the NND on environmental impact was that import distances were not included in the present study.

The NND studies (Saxe 2013, Saxe 2014, and the present study) have shown that the diet composition, the meat quantum and meat type ratio, the transport distance of imported commodities and the inclusion of organics all affect the environmental impact of what we eat and drink. So we asked ourselves if there are *other* factors which could affect the environmental impact of a diet. The modeled substitution of soft drinks with a designer drink in the SW-ADD proved that in theory, manipulating with the sensory quality of what we eat and drink can affect our sense of satiety and thus make us eat less. Eating less effectively saves the consumer unnecessary expenses and saves society and citizens environmental impact and thereby reduces the shadow price of diets. Since two billion people worldwide are overweight or obese, a sensory improvement of our diets would not only increase the palatability, but also improve our health - by "manipulating" consumers to eat less.

There are indications that a high-protein diets are more satisfying (e.g. Padon-Jones et al. 2008), which may also be a way to decrease our intake, and thereby improve the general health and the environmental impact of diets.

5. Conclusion

This study supports the findings by Saxe (2014) that the OPUS NND is a surprisingly efficient tool in environmental protection – even when modeled in a "real-life" scenario based on expected consumer preferences; an instrument that can be further tuned and refined.

The increased consumer cost (284 €/capita/year) associated with an ADD-to-NND diet-shift is only partly (70 %) countered by the reduction in environmental costs (€151/capita/year) associated with the NND. Therefore, only if the health benefits of the ADD-to-NND diet-shift is at least half of the environmental benefits, will it be a socioeconomic advantage to society if consumers prefer the New Nordic Diet over the Average Danish Diet. The potential savings associated with the reduced environmental impact of consuming the NND rather than the ADD is significant, as it equals a quarter of the environmental impact of driving a modern passenger car per capita and year.

The fact that the shadow price of a diet's meat ingredients, in contrast to the shadow price of all other ingredients was found to be higher than the consumer price, supports that a regulation of meat prices would be of singular importance for politicians and legislators when focusing on in future environmental regulation. The consumer should pay for the environmental (and health) impacts inflicted through their diet choices – no more, no less. That is the way of regulating the prices of automobiles, heat and power in Denmark. So why not apply the same instrument for food?

Though reducing the meat content in a diet seems to be the most efficient way to reduce the environmental impact of eating and drinking, there are obvious alternatives. One is to substitute a proportion of red meat with white meat, even when keeping the meet content constant. Another alternative is to eat less, either induced by our own free will (e.g. to "get in shape") or by seducing us to eat less through a higher protein content in our diet, or because of a higher sensory satisfaction e.g. via content of *umami* in your diet. Other alternatives include buying more local and less imported produce (which is true for most but not all commodities, Saunders et al. 2006), and overall buying less organics (or better, focusing on the environmentally friendly organics). The latter statement is based on comparing the monetized environmental impact of a range of organic vs. conventional products (Saxe 2014). Most organic produce has a higher overall environmental impact than their conventional counterpart.

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Future Atmospheric CO_2 Concentration and Environmental Consequences for the Feed Market: a Consequential LCA

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ABSTRACT

With the rising atmospheric carbon dioxide concentration [CO_2], crops will assimilate more carbon. This will increase yields in terms of carbohydrates but dilute the content of protein and minerals in crops. This consequential life cycle assessment study modelled the environmental consequences that such altered chemical composition and crop yields would have for the production of pig feed. Results revealed, among others, that an extra European demand of pig feed under an atmospheric [CO_2] of 550 µmole mole^{-1} would lead to ca. 6% less expansion of additional arable land worldwide, in comparison to feed produced under today's conditions. However, this did not translate into lower greenhouse gas emissions, because the benefit of increased crop yield was counteracted by changes in the composition of the feed formulation. Among the important changes, feed produced under high [CO_2] was shown to integrate 23% more soymeal and 5% less wheat than at present.

Keywords: Compound pig feed formulation, consequential life cycle assessment, land use changes, amino acids.

1. Introduction

During this century, the rising atmospheric carbon dioxide concentration ([CO_2]) is expected to increase crop yield due to increased carbon assimilation, for example in wheat grains by 10% (Högy et al. 2009). On the other hand, the higher carbon uptake causes increased starch content and, consequently, also a dilution of the relative protein and mineral content in food and feed crops.

The animal feed market, which currently absorbs approximately 45%, 58%, and 80% of the world cereal, maize and soy, respectively (International Grain Council 2012), is likely to be affected by such changes in crop yield and composition. In fact, as the composition of animal feed for intensive livestock production is carefully optimized in order to meet both the applying legislation and the nutritional requirements in a cost-effective manner, any change in both the chemical composition and the yield of feed crops will affect the future formulation of the feed.

Although the environmental consequences of the forecasted future increases in meat (and thus feed) demand have been overly studied (e.g. Steinfeld et al. 2006; Nellemann et al. 2009; Meul et al. 2012), there are no studies, to the authors' knowledge, that attempted to address the environmental impacts related to the changes that a higher [CO_2] would induce to the feed market. On an environmental perspective, a (CO_2-induced) higher crop yield would contribute to reduce both the land use and the land use changes associated with increased demand for animal feed. On the other hand, lower protein content would trigger an increased need for protein-rich crops like soybeans, which in turns would involve an increased land use change, the environmental impact of which may be considerable (e.g. Searchinger et al. 2008; Gibbs et al. 2008; Hamelin et al. 2014). Yet, it is not clear which effect would dominate over the other. Further, a change in protein content has to be addressed with regards to the specific amino acids affected, as a change in some non-essential amino acids (e.g. proline) is rather meaningless for the overall feed composition.

Focusing on the effects of elevated [CO_2] alone, this study endeavours to assess the environmental consequences of compound pig feed grown at the "present" atmospheric [CO_2] (taken at 380 µmole CO_2 mole^{-1} dry air) and at a "future" [CO_2] (550 µmole mole^{-1} dry air by 2050; IPCC 2007). From this point onwards, the former will be referred to as "present feed", and the latter, as "future feed" with the understanding that these are both compound feeds. Compound (or complete) feed means multi-component feed produced at a competitive price to satisfy all nutritional and technical needs in terms of carbohydrates, protein, oils, fibre, enzymes, vitamins, etc.

2. Methods

2.1. Feed Formulation

For a market-based approach, the feed formulation software Bestmix® (Adifo Software 2014) was used to establish both the present and future feed formulation for piglets, sows and slaughter pigs (a feed mix consisting of 20% piglet feed, 20% sow feed and 60% fattening pig feed was considered). Through optimization algorithms, Bestmix® formulated the best compromise between profitability, nutritional value and animal health, while ensuring that the Danish and European legal requirements of the feed are met. This procedure allowed determining the exact ingredients constituting the present and future pig feed, and in which proportion.

One important input to Bestmix® is the chemical composition of crop ingredients, in terms of starch, protein, amino acids and macro- and micro-nutrients, for both today's and future's conditions (Table 1). Another important input parameter to Bestmix® is the price of feed ingredients. However, as the aim of this study is primarily to investigate the consequences of crops' composition changes, and as no reliable estimate is available on future crop prices, prices of crop ingredients under a high [CO_2] future were taken to be equal to the present prices.

2.2. LCA Model

The functional unit to which all input and output flows were related is "the production of one extra tonne of pig feed". Background LCA data (e.g. fertilizers, electricity, etc.) were taken from the Ecoinvent v.2.2 database (Ecoinvent Centre 2014), while foreground data were essentially related to the cultivation, yield and chemical composition of the crops used in the feed formulation. The geographical scope considered for the foreground system was Denmark, i.e. the pig feed was considered to be manufactured for the Danish market. The Danish example is taken as representative for most industrialized countries in a temperate climate. The chosen method of normalization was to monetize the environmental impacts (expressed in €2003; Stepwise2006 v1.2 impact assessment method, Weidema 2009). Although the results from this step are not presented here, this allowed the selection of the key characterized impact categories for this study; global warming (100y horizon), nature occupation, respiratory inorganics and non-carcinogenic human toxicity.

All processes affected by the production of 1 tonne of pig feed, i.e. from crop cultivation to harvest, and up to the mixing of the feed were included in the LCA system boundary, including, among others, the cascading effects induced by a change in the biochemical composition of the ingredients. This change implies an overall change in the feed formulation, both in terms of the ingredients employed and in their quantities in the feed, which was quantified though Bestmix® (output of Bestmix®; Table 2).

2.3. System Boundary of Crop Ingredient and Life Cycle Inventory (LCI)

This study involves 5 main crop ingredients, and for all these, land use changes were taken into account. The wheat needed for the pig feed considered in this case study is grown in Denmark. In a country like Denmark, where 65% of the total land is already used for cropland and where policies have been adopted in order to double the forested area (nowadays representing ca. 13% of the total land; Nielsen et al. 2011), very limited conversion from forest or alike nature types is occurring. Most likely, the land needed to grow this extra wheat will be taken from actual Danish cropland, involving that one crop cultivated today will be displaced. Such a displaced crop is, in consequential LCA, referred to as the marginal crop. In this study, the marginal crop was assumed to be spring barley (a carbohydrate crop), based on Schmidt (2007) and Dalgaard et al. (2008). The environmental consequences of cultivating wheat instead of the spring barley that was already cultivated represent, in this case, the so-called direct land use changes (dLUC).

Table 1. Chemical composition and yield of the four main crop ingredients, at present (380 µmole mole^{-1}) and future (550 µmole mole^{-1}) atmospheric CO_2-concentration

	Wheat, Denmark			Barley, Denmark			Soy, Argentina/Brazil			Rapeseed, Germany		
Chemical composition	Present[1]	Change[2] (%)	Future	Present[1]	Change[3, 4] (%)	Future	Present[1]	Change[3] (%)	Future	Present[1]	Change[5] (%)	Future
Total Dry Matter, %	85	0	85	85	0	85	88	unknown	unknown	89	unknown	unknown
Starch (g/kg)	670	+7.5	720	630	+4.1[4]	660	27	unknown	unknown	19	unknown	unknown
Raw Protein, %	9.5	-14	8.2	9.7	-15[3]	8.2	47	-1.4	46	35	-4.6	33
Amino Acids (g/kg)												
Lysine	2.3	-10	2.0	2.6	unknown	unknown	26	unknown	unknown	15	-2.5	14
Methionine	1.3	-17	1.1	1.4	unknown	unknown	5.7	unknown	unknown	6.0	-4.3	5.7
Cystine	1.9	-17	1.6	1.8	unknown	unknown	5.8	unknown	unknown	7.0	-5.4	6.6
Threonine	2.2	-21	1.8	2.4	unknown	unknown	16	unknown	unknown	12	-1.8	11
Tryptophan	0.93	-17	0.77	0.93	unknown	unknown	5.7	unknown	unknown	3.3	-3.1	3.2
Isoleucine	2.8	-22	2.2	2.7	unknown	unknown	18	unknown	unknown	11	-3.1	10
Leucine	5.2	-19	4.2	5.1	unknown	unknown	32	unknown	unknown	20	-3.7	19
Histidine	1.9	-16	1.6	1.7	unknown	unknown	11	unknown	unknown	8.1	-3.9	7.7
Phenylalanine	3.6	-14	3.1	3.7	unknown	unknown	21	unknown	unknown	11	-1.9	11
Tyrosine	2.4	-16	2.0	2.3	unknown	unknown	16	unknown	unknown	8.2	-2.6	8.0
Valine	3.5	-8	3.2	3.6	unknown	unknown	20	unknown	unknown	14	-3.4	13
Macro-elements (g/kg)					(4)							
Calcium	0.43	-15	0.36	0.43	4.8	0.45	3.5	unknown	unknown	8.5	-1.5	8.4
Phosphorus	2.6	-4	2.5	3.0	-4.6	2.8	6.8	unknown	unknown	11	1.8	11
Sodium	0.085	-6	0.080	0.17	unknown	unknown	0.18	unknown	unknown	0.36	0.1	0.36
Potassium	5.0	-1	4.9	4.8	0	4.8	22	unknown	unknown	13	0.3	13
Magnesium	1.0	-7	0.95	1.0	-3.3	0.99	3.2	unknown	unknown	4.7	0.8	4.8
Sulphur	1.1	-13	0.96	0.94	-5.0	0.89	3.6	unknown	unknown	6.5	-6.1	6.1
Micro-elements (mg/kg)					(4)							
Iron	27	-18	22	29	-11	26	260	unknown	unknown	230	1.0	230
Manganese	25	-3	24	12	unknown	unknown	47	unknown	unknown	68	-2.6	66
Zink	34	-13	30	24	-13	21	48	unknown	unknown	65	-5.6	61
Yield (t fm ha^{-1})	6.8[6]	+ 11%[5]	7.3	5.0[6]	+20%[8]	6.0	3.4[9]	+15%[10]	3.9	3.8[11]	+ 5.0%[12]	4.0

[1] VSP 2012; [2] Högy and Fangmeier 2008; [3] Taub et al. 2008; [4] Erbs et al. 2010; [5] Högy et al. 2010 (extrapolation assuming linear responses to elevated [CO_2]); [6] Hamelin et al. 2012; [7] Högy et al. 2009 (extrapolation assuming linear responses to elevated [CO_2]); [8] Fangmeier et al. 2000; [9] Dalgaard et al. 2008; [10] Morgan et al. 2005; [11] FAOSTAT 2014 (average 2005-2010); [12] Clausen et al. 2011.

Indirect land use changes (iLUC), on the other hand, represent the environmental consequences related to how this missing supply of Danish spring barley will be supplied on the world market. Such increased crop production may stem from increased yield, also referred to as intensification, or from land conversion to cropland, also referred to as agricultural land expansion. Both were taken into account, as further described in 2.4.

Barley, rapeseed meal and sunflower meal were also considered to be imported from Europe. Similarly to wheat, an increased demand for land in Europe in order to cultivate the barley, rapeseed (rapeseed meal) and sunflower (sunflower meal) resulting from an extra demand of Danish pig feed was considered to take place at the expense of spring barley. In the case of rapeseed and sunflower, the meal needed for the feed is however co-produced with respective rape and sunflower oil, leading to a corresponding decrease production and supply of the marginal oil, here taken as palm oil (Schmidt 2007; Dalgaard et al. 2008). In terms of land use changes, this represents an avoided cultivation from the marginal supplier of palm oil, here considered to be South-East Asia, where the largest increases occurred since the mid-1960s, as highlighted by the production statistics from the Food and Agriculture Organization (FAO) of the United Nations (FAOSTAT 2014). Yet, along with this palm oil, palm meal would have been produced as well (Schmidt 2007). As a result of the no longer produced palm meal (supplying both carbohydrates and protein), the cultivation of the marginal source of carbohydrate and protein is induced, here taken as Canadian barley (Schmidt 2007) and soybean meal (Dalgaard et al. 2008), respectively. Because the production of the marginal protein, soybean meal, interacts with the oil market again, a loop system is thus created, and this loop should be stopped at the point where the consequences are so small (i.e. when the differences between two subsequent iterations approach zero), that any further expansion of the boundaries would yield no significant information for decision support (Ekvall and Weidema 2004). These cascading effects are referred to as the "oil-meal loop". The substitution ratios were quantified on the basis of the carbohydrates and lysine content of the displaced and induced/avoided crops. The content in lysine was used instead of the crop’s content in total protein; it is in fact the composition of the protein in terms of amino acids, or rather in terms of limiting amino acids, which matters for feed, and lysine is the most important limiting amino acid in pig feed.

For soybean meal, based on an analysis of the historical data available in the statistical database of FAO (FAOstat, 2014), soybean meal from Argentina and Brazil was identified as the one most likely to react to an increase in demand for soy. For palm fatty acid distillates (palm fruit), as above-mentioned, the palm meal from South-East Asia was considered. The system expansion considered for these two oil crops is as described above, both involving the oil-meal loop.

For fermentation-based amino acids, which are produced from of a mix of different crops, the same principles as described above were applied to define the system boundary.

The inventory data for barley and wheat cultivation in Denmark were taken from Hamelin et al. (2012; wet climate, sandy soil), while life cycle inventory (LCI) data from the Ecoinvent (v.2.2) database were used to model the cultivation of imported soybean (Brazil), rapeseed (Germany), sunflower (Spain) and palm oil (Malaysia). For fermentation-based amino acids, the generic recipe of (Mosnier et al. 2011) was used, while all enzymes and vitamins were modelled as phytase, this being the best proxy found given the availability of LCI datasets at the time of modelling. For other ingredients (mineral, salts, fish meals), LCI datasets from the Ecoinvent (v.2.2) database were used.

2.4. Land Use Changes

In this study, the iLUC impact was defined as the sum of: 1) net arable land expansion, 2) intensification, and 3) cultivation of the reacting crop on the new agricultural land. To quantify this impact, it is necessary to identify: i) which regions are likely to be affected; ii) in these, which types of biomes are converted; iii) in each affected region, how much land is affected and how (i.e., expected share of intensification and expansion); iv) C losses from the converted land; v) changed use of N-fertilizers associated with intensification; vi) cultivation practices for the crops established on the cleared land.

To this end, the approach described in (Hamelin et al. 2012b) was applied. The reader is referred to this, and to Hamelin (2013) and Tonini et al. 2012, for additional details on the modelling of land use changes.

3. Results

Table 2 presents the optimization output performed in Bestmix®, i.e. the composition of present and future compound pig feed based on the biochemical composition of the main crop ingredients (Table 1) and the current price of these. The results in Table 2 support the hypothesis that pig feed based on wheat, barley and rape grown in Northern Europe under higher [CO_2] will need a higher supplement of protein as well as less carbohydrate ingredients. Table 2 also highlights that both present and future feed consist of at least 46% wheat, 25% barley, 9% soy meal, 7% rape meal and 4% sunflower seeds, these five ingredients making up approximately 93% of the feed.

Characterized LCA results are presented in Table 3, with focus on the four socio-economically most important impact categories identified through normalization in Stepwise 2006, as described in 2.2.

Table 2. Change in ingredients of compound pig feed under a future [CO2] (550 ppm) (output of Bestmix®)[1]

Product	Main function in the feed	Amount in compound pig feed		
		present ($kg\ t^{-1}$ feed)	future ($kg\ t^{-1}$ feed)	relative change
Wheat	Energy	479.9	455.2	-5.1%
Barley	Energy	250.0	250.0	0.0%
Soy meal	Protein	92.2	113.7	+23%
Rape meal	Protein	68.0	68.0	0.0%
Sunflower meal	Protein	42.0	42.0	0.0%
Beet molasses	Energy	17.0	20.0	+18%
PFAD[2] oil	Energy and technical aid	13.4	13.0	-2.9%
Chalk (lime) $CaCO_3$	Health	12.0	11.7	-2.1%
Amino acids (fermentation[3])	Health	5.1	4.3	- 16%
Salt, sodium chloride	Health	4.5	4.5	+ 6.3%
Mono calcium phosphate	Health	4.3	4.7	+ 5.5%
Protein (from fish)	Protein	4.0	4.0	0.0%
Vitamins	Health	3.2	3.2	0.0%
Phytase and xylanase	Health, economy, environment	0.8	0.8	0.0%
Dl-methionine (synthetic)	Health	0.2	0.3	+ 14%
Haemoglobin meal	Protein	2.0	3.4	+ 65%
Formic acid, calcium salt	pH-adjustment	1.16	1.16	0.0%

[1]The number of digits does not reflect the precision; these are simply shown in order to facilitate the comparison between the different feed ingredients; [2]Palm fatty acid distillate; [3]Includes lysine, threonine and tryptophan.

Table 3. Characterized LCA results for the most important impact categories, as identified by the monetized normalization in Stepwise, split up per feed ingredient[1]

	Non-carcinogenic human toxicity (kg C_2H_3Cl-eq. per t pig feed)		Respiratory inorganics (kg $PM_{2.5}$-eq. per t pig feed)		Nature occupation (m^2 agricultural land per t pig feed)		Global warming (kg CO_2-eq. per t pig feed)	
	Present	Future	Present	Future	Present	Future	Present	Future
Amino acids (fermentation)	-3.62	-2.44	-0.046	-0.030	-117	-76	-67	-45
Barley	2.74	2.50	0.079	0.072	172	157	162	150
Palm oil	-0.76	-0.64	0.088	0.075	-0	-0	51	43
Rape	30.40	24.17	-0.220	-0.193	140	119	-4	-14
Soy	-6.61	-7.15	0.375	0.418	185	202	409	450
Sunflower	-7.59	-6.09	-0.105	-0.086	478	406	42	44
Wheat	31.12	27.16	0.200	0.182	258	243	-9	13
Others	0.10	0.10	0.018	0.019	1	1	30	30
Net	*45.78*	*37.51*	*0.389*	*0.457*	*1117*	*1051*	*613*	*671*

[1]The number of digits does not reflect the precision; these are simply shown in order to facilitate the comparison between the different feed ingredients.

4. Discussion

The environmental impact of producing one extra tonne of compound pig feed was expected to decrease in the high [CO_2] future due to the higher yields (and thus lower land use) caused by the higher CO_2 uptake from the atmosphere. But even though the yield increases were found to be non-negligible, e.g. 10% for wheat and 28% for rape (Table 1), the net land use did not decrease to the same extent. Still, however, the net fall in land use per tonne of pig feed was found to be around 6% (Table 3). Unexpectedly, this did not lead to a net fall in greenhouse gas emissions per tonne of feed, on the contrary it increased by 9% for crops grown at the future [CO_2]. This is essentially due to the change in feed composition, where considerably more soybean meal is required under a high [CO_2] (23% more; Table 2), which involves the conversion of biomes with relatively high C stocks.

The biggest differences, however, are found for the impact categories respiratory inorganics (fine particles) and non-carcinogenic human toxicity (17% increase and 18% decrease, respectively). Again, these differences between the present and future [CO_2] scenarios can be explained by the change in feed's chemical composition, especially the increased need for soy meal and decreased need for wheat (Table 2).

For all impact categories, nearly all impacts are caused by the crop-based ingredients, i.e. the sum of others (non-crop ingredients) is infinitesimal (Table 3). For non-carcinogenic human toxicity, rape and wheat are the biggest contributors (caused by the agrochemicals used for rape and wheat cultivation, but for wheat much reduced by the displaced barley). For respiratory inorganics, soy and wheat are the major contributors (caused by soy and wheat cultivation, but for wheat much reduced by the displaced barley, and for soy somewhat reduced by soy oil displacing palm oil), while rape contributes negatively (mainly due to displaced palm oil). For nature occupation all the main crop ingredients contribute, and sunflower the most (due to the low yield of sunflower cultivation and the displaced barley). For global warming, soy is the biggest overall contributor (essentially because of iLUC), followed by barley. Amino acids produced by fermentation contribute negatively to each of the main impact categories. This is because sugar production (one of the substrates in the fermentation process producing amino acids) gives rise to by-products (molasses and pulp) which can substitute the use of marginal carbohydrates for animal feed. The saved carbohydrates production (and the land use changes it would have generated) had a greater negative impact than the positive impact from the consumed sugar substrate. Yet, these effects are of course highly dependent upon the data quality used to model them. As three crop ingredients are used to produce these amino acids (sugar beet for the sugar input, corn for the corn starch input and wheat for the wheat starch input), and as each of these crops involve at least three co-products, a considerable degree of uncertainty is introduced in the model, as a result of the numerous assumptions involved regarding the displacement effects (i.e. system expansion).

One limit of this study is that it only focussed on the effect of elevated [CO_2] to represent the future conditions under which pig feed will be produced. In fact, higher atmospheric [CO_2] is not the only global change to determine the yield and the chemical composition of crops in the future. The results of this study are thus not to be seen as representative of the future state of global climate and meteorological conditions (data for this is as yet unavailable), but as an illustration of the cascading consequences a change in crop yield and composition (in this case triggered by an increase in atmospheric [CO_2]) can have for pig feed formulation. As for any LCA, the results of this study are closely linked to the quality of the inventory data and assumptions taken. For example, no changes in amino acid composition were considered for barley and soy as a result of increased [CO_2], because no data or reliable estimation proxy could be found. Nevertheless, the study provides a solid framework for assessing the consequences a changed crop composition due to elevated atmospheric [CO_2] would have on pig feed, which was not, to authors' knowledge, available so far.

Another limit is that the changes in manure composition resulting from a change in feed composition have not been taken into account. In fact, a feed containing less protein from cereals (which are difficult to digest) but more easily digestible protein from e.g. soy or rapeseed meal, involves a better digestion, and thus a reduction in excreted N can be expected. This could have consequences for the subsequent use of the manure as a fertilizer, as it would involve a reduced potential for emission of N flows (e.g. ammonia, nitrous oxide, nitrate losses). Based on Table 2, considering these induced changes in manure composition would likely have induced additional benefits for the future pig feed, which comprises significantly more soymeal and less wheat.

Finally, one of the most important sources of uncertainty probably lies in the estimation of the environmental consequences generated by land use changes, as clearly emphasized in several publications already (e.g. Plevin et al. 2010; Warner et al. 2013). Nevertheless, it should be highlighted that although the actual magnitude of environmental impacts related to land use changes is uncertain, the potentiality of adverse effects arising from it is hardly subject to dispute (Marelli et al. 2011; Khanna and Crago 2012).

5. Conclusion

The main findings and highlights of this study can be summarized as follows:

- A methodological framework was presented in order to assess the cascading environmental consequences a change in crop yield and chemical composition (triggered by an increase in atmospheric [CO_2]) can have for pig feed formulation, including land use change consequences;
- Due to changes in biochemical crop composition, pig feed grown under an elevated [CO_2] would contain more soy (ca. 20%) and less wheat (ca. 5%), in comparison with today's feed.
- The positive environmental effect of elevated [CO_2] on crop yield (carbohydrates) was counter-balanced by a need for increased soy content in pig feed, and the land use consequences this generated;
- The four most important environmental impact categories in pig feed production under current and future atmospheric [CO_2], as determined by the monetized normalization methodology of Stepwise 2006 were human toxicity in terms of non-carcinogenic toxicity and respiratory inorganics, nature occupation and global warming;
- Since the protein crops (soy, rape and sunflower) account for about 60% of the overall environmental impact of pig feed, it is important, in the perspective of a future with expected growing demands for food and bioenergy (and thus for land), to optimize their supply in feed as well as their amino acids profiles (rather than their content in total protein).

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Comparing global and product-based LCA perspectives on environmental impacts of low-concentrate ruminant production

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ABSTRACT

Results of LCA studies from single supply chains or of specific contexts are often generalized and transferred to a global perspective. The aim of this paper is to compare results gained from a global land use and mass-flow model and a consistent set of assumptions with product-related attributional LCA results. We apply the model to question of livestock intensification strategies – in particular with respect to the use of human-edible feedstuffs in ruminant rations. From a resource efficiency perspective of attributional LCAs, our results show that concentrate-based ruminant production outperforms grass-based production. From a global perspective, however, a reduction of concentrate feedstuffs could reduce environmental impacts of the total sector while producing the same amount of human-edible energy and protein. Dietary patterns, however, would have to change towards less meat consumption. We conclude that if aiming at drawing conclusions with large-scale relevance, a global perspective needs to complement an efficiency-centered LCA perspective.

Keywords: food security, physical mass-balance model, dietary change, greenhouse gases, attributional LCA, consequential LCA

1. Introduction

Life cycle assessment is a useful approach for comparing different products or production methods to each other, because LCA relates all environmental impacts to a functional unit. As this functional unit mostly refers to the food production, food availability is implicitly addressed in the assessment. However, contrary to many other economic sectors, in agriculture there are absolute natural boundaries for growing specific crops, complex interdependencies between crop and livestock production, and differences between site-specific climatic and soil conditions (Rockström et al., 2009; Stoorvogel, 2014). Non-linearity can only be covered in a limited way, by an LCA creating a ratio between environmental impacts and a functional unit.

The environmental impacts of livestock production are frequently evaluated using carbon footprints or life cycle assessments including the assessment of different environmental impacts (Basset-Mens und van der Werf, 2005; Steinfeld, 2006). Especially the question of livestock intensification is often addressed (e.g. Gerber *et al.*, 2011). The frequent conclusion is increasing milk yields mostly over-compensate additional environmental impacts of feed production (Nguyen *et al.*, 2013). Besides breeding, concentrate feed is the important factor in such an intensification strategy The impacts of concentrate feed is particularly relevant for greenhouse gases of ruminants, as with longer fattening periods, more methane is emitted (Gerber *et al.*, 2013) through enteric fermentation. Thus, from a single-supply-chain perspective, we can conclude that the use of concentrates should be favorable for the environment.

However from a global perspective, two thirds of the global agricultural area is covered by grasslands (FAOSTAT, 2013). A large share of this grassland cannot be converted to productive arable land either due to slopes, soil characteristics or due to rainfall patterns (Suttie *et al.*, 2005). Using crops grown on arable land increases pressure on a globally scarce resource. So, does it makes sense to increase livestock productivity by increasing shares of human-edible feedstuffs and crops grown on arable land in feeding rations or alternatively rather reduce this share? Analyzing such a fundamental question requires an approach taking into account the absolute boundaries in availability of natural resources from a global perspective.

In this paper, we aim to present a new LCA-based approach for analyzing the impacts of fundamental changes in the global food system. We apply this approach for analyzing the impacts of a reduction in concentrate feed use for ruminant production compared to a reference scenario for 2050. We compare the results on resource efficiency per kg of milk with results at global level, using the amount of energy available for human nutrition as a global functional unit. This allows us not only to analyze at resource efficiency but also consistency and efficiency as two complementary sustainability strategies (Schaltegger *et al.*, 2003). Finally, we draw conclusions about the explanatory power of single-supply chain LCAs with respect to global environmental challenges.

2. Methods

To analyze the environmental impacts of global food system scenarios, we used the global model SOL-m (Schader et al., 2012) and further developed it to a LCA database for global land use activities. The model builds upon FAOSTAT, including the food balance sheets, tradestat, fertistat and other databases (FAO, 2013). Further data sources for life cycle inventories were based on LCA databases (Nemecek und Kägi, 2007) and additional scientific literature and datasets (e.g. GLC2000 data transformed according to Erb *et al.*, 2007). Plant production (180 activities) and livestock production (22 activities) are defined for 229 countries. Activity and country-specific defined inputs and outputs allow to model environmental impacts and to aggregate them to regional or global level (Figure 1). The model is programmed in GAMS (General Algebraic Modeling System).

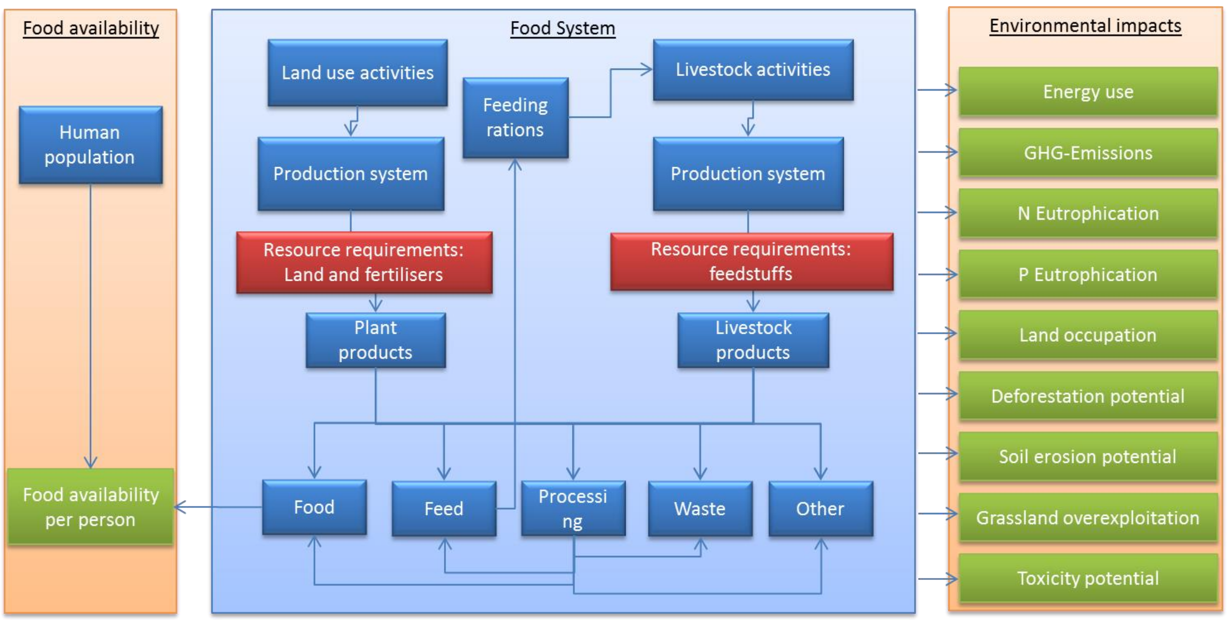

Figure 1. SOL-m model structure with food availability and environmental impacts as model outputs

Global food availability is calculated according to Equation 1, summing up the outputs in terms of mass, energy and protein used for food for all activities, production systems and geographic units.

$$GlobalFoodAvailability_{i,m} = \sum_{ijk} AL_{i,j,k} * OUT_{i,j,k,Yield,Mass} * NCHC_{i,j,k,m} * UF_{i,j,k,food} \forall_{i,m} \quad (1)$$

Where i = index of geographic units;
j = index of activities;
k = index of farming systems;
l = index of inputs and outputs;
m = index of nutrients for human consumption;
n = index of utilization types;
s = index of units of inputs and outputs;
FA = food availability [kcal or g protein]
AL = activity level [ha/year for land use activities, number of animals/year for livestock activities]
OUT = output [kg/ha or kg/animal]
NCHC = nutrient contents for human consumption [%]
UF = utilization factor [%]

The model covers the environmental impacts land occupation (differentiated by arable and grassland), energy use, greenhouse gas emissions, from a life cycle perspective, based on Equation 2. Greenhouse gas emissions were calculated using the method GWP_IPCC100a. For agricultural emissions a Tier 1 approach was used, except for methane emissions from enteric fermentation for which we used a Tier 2 approach (IPCC, 2006). Allocation procedures can be switched between mass, energy, protein and economic allocation. N and P-surplus are modelled correspondingly but taking into account only nutrient flows in the agricultural system.

$$Environmental\ impact_{i,o} = \sum_{ijk} AL_{i,j,k} * (IN_{i,j,k,l,s,o} + OUT_{i,j,k,l,s,o}) * IF_{i,j,k,l,s,o} \quad (2)$$

Where o = index of environmental impacts:
IN = inputs [kg]
IF = impact factors [environmental impact / kg of input]

For each activity, we defined inputs and outputs, i.e. all relevant physical flows related to individual activities. Inputs for livestock activities include four categories of feedstuffs: a) forage from crops grown on arable land, b) concentrate feed (grains, pulses grown on arable land), c) grassland-based fodder and d) agricultural/agri-industrial by-products. Further inputs for livestock activities are energy input for buildings, in-stall processes and fences. While a) and b) are in competition with production of human-edible food, c) and d) are not. Outputs of animal production activities include human-edible products (meat, milk, eggs, honey) plus hides and wool, manure excretion, nutrient losses and GHG emissions due to enteric fermentation and manure management (CH_4, N_2O, NO_3, NH_3). Amounts of concentrate feed and by-products subsume country-specific feed amounts according to the food balance sheets from FAOSTAT. Inputs for plant production activities included arable or grassland areas, mineral fertilizers (N, P, K), manure, crop residues, symbiotic nitrogen fixation, herbicides, fungicides, insecticides and management practices (tillage, seeding, fertilization, spraying, irrigation, flooding of paddy rice, harvesting, transport and drying). Outputs from plant production activities include crop yield quantities, crop residues and nitrogen losses during fertilizer application.

Thus, main drivers (yields, fertilizer and nutrient balances) of environmental impacts could be covered based on country and crop-specific data. The model can thus illustrate impacts of major changes in production patterns, processing, food waste and food demand and serves to understand possible interdependencies.

We model pesticide use potential, freshwater use for irrigation, annual deforestation pressure and soil erosion potential as additional environmental indicators. The pesticide use intensity model takes into account three parameters: pesticide intensity levels of crops, pesticide legislation in a country and access to pesticides by farmers. Parameter are expressed on a semi-quantitative scale and multiplied with each other. Freshwater use for irrigation is based on AQUASTAT datasets and covers estimations by crop and country. Annual deforestation pressure is derived from additional arable land and grassland required for agricultural production. We calculated an average annual deforestation rate for the years 2005-2009. Soil erosion is based on soil loss rates (Schwegler, 2014) and soil erosion susceptibility of crops as a function of the period of time when soil is left bare.

Apart from global scenarios, the model allows to show results by country or region respectively. Also crop and livestock type-specific data can be displayed. This allows comparing the resource use of average crop or livestock production activity for the functional units of a) mass, b) human-digestible energy produced, c) human-digestible protein produced, d) revenues generated and e) per hectare of land and year. The possibilities for presenting results are only limited by lacking or low-quality data, particularly on production systems in developing countries.

The model was calibrated to a base year which is constituted by the arithmetic mean of 2005-2009. Agricultural greenhouse gas emissions were compared with FAOSTAT emissions database (Tubiello *et al.*, 2013) and non-agricultural emissions for inputs were mainly taken from ecoinvent (Frischknecht *et al.*, 2007).

The besides the base year, the following scenarios were calculated for 2050:

Reference scenario: The reference scenario was developed according to Alexandratos and Bruinsma (2012), which forecast the global population increase, food demand patterns, land use patterns, livestock numbers and technical progress in terms of crop and livestock yield increases. Potential technical progress in the energy sector (e.g. increases in renewable energy use in the national energy mixes) was not considered.

Zero-concentrate ruminant production: This scenario builds basically on the same assumptions as the reference scenario, with respect to population increase, land use patterns, and technical progress. However, the use of human-edible concentrates and forage crops grown on arable land is minimized. This means that only grass and shrubs as well as plant-based waste from the food industry are used. The area of permanent grasslands was fixed and ruminant per head productivity was assumed to reduce by 0-40%, as there is clear reliable data about potential reductions on potential yield reductions. In this paper, we show only the results of the medium scenario which assumes on average 20% of yield reduction due to concentrate reduction. Livestock numbers and dietary patterns, however, are modelled as an endogenous variable in the model by looking at the bio-physical planetary boundaries.

3. Results

Figure 2 shows the impacts of a reduction in human edible concentrate use on global warming potential (GWP) per kg of milk produced. Regions in which a low average milk yield is dominant (Africa, South Eastern Asia, Latin America) show a high global warming potential as compared to regions with intensive dairy production (Europe, North America, Western Asia). Furthermore, the reduction of human-edible concentrates in ruminant rations results in an increase in GWP by 25-88% compared to the reference scenario. At global level, the impact categories total land occupation (+91%), deforestation potential (+95%), N-surplus (+20%), P-surplus (+59%), and soil erosion potential (+67%) increase per ton of milk. However, decreases can be shown for energy demand (-52%), freshwater use (-62%), occupation of arable land (-100%) and pesticide use intensity (-100%).

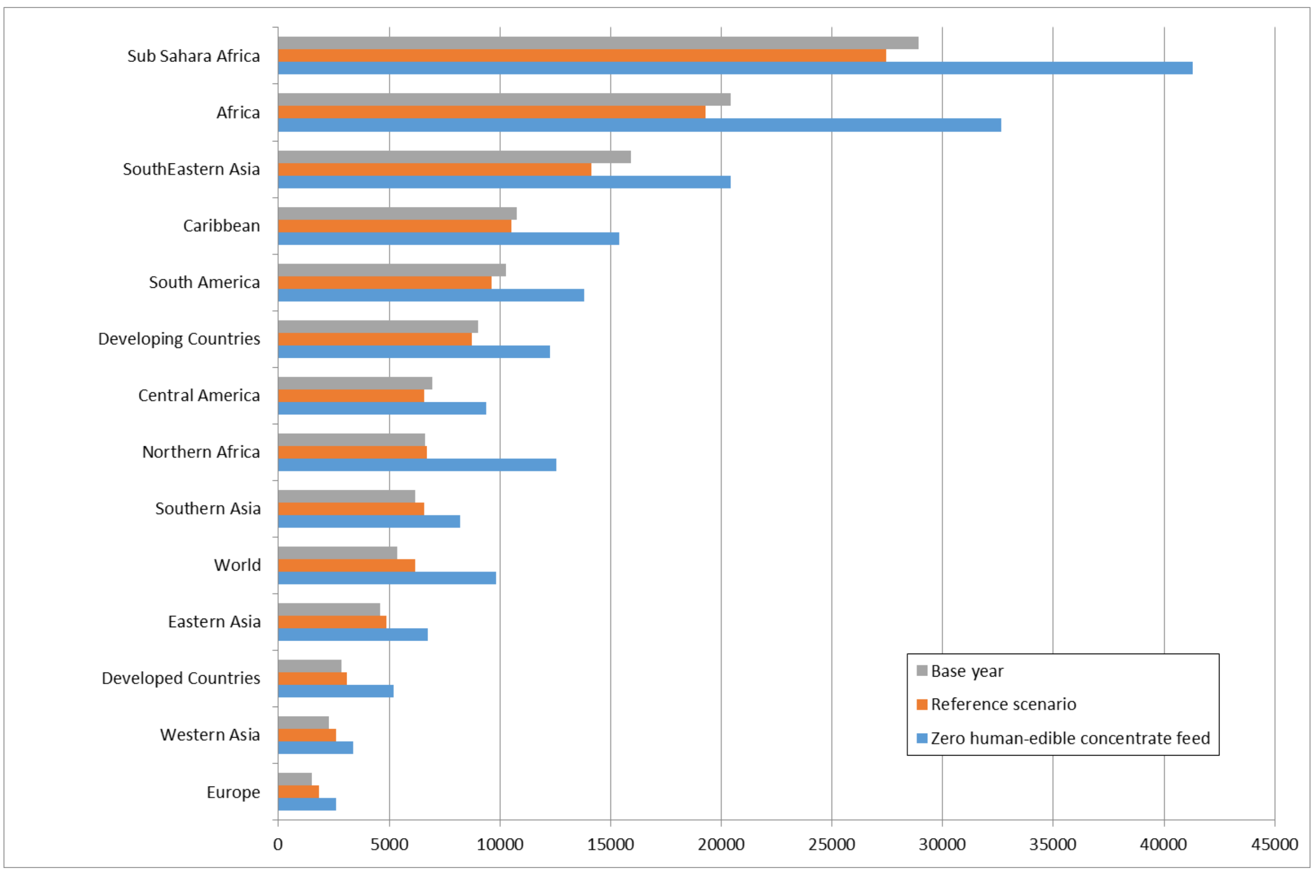

Figure 2. Comparison of average global warming potential [kg CO2-eq] in different regions per t milk delivered by dairy cattle

In Table 1, the impacts of the reduction in human-edible concentrate use on meat and milk availability are presented. As a consequence of reducing human-edible feed, the occupation of agricultural land decreases by 2.6% compared to the reference year, as the arable land goes down by 8.4%. This is a direct result of the lower amount of concentrates and forage crops that are cultivated on arable land. However, livestock yield reductions and lower cattle numbers partly compensate this effect. Cattle numbers will have to decline by almost 32%, buffaloes, goats and sheep numbers only decline by 2-6% due to lower shares of concentrates in current rations.

The share protein in human diets which is delivered of animal products goes down by 17% (4% compared to the base year), the energy shares delivered by animal products go down by 20%, compared to the reference scenario (6% compared to the base year).

This results in a 12% reduction in GWP. Other environmental impacts go down by 2% (freshwater use) to up to 20% (N-surplus).

Table 1 Impacts of a global reduction of concentrate feed for ruminant production on environmental impacts, agricultural production, dietary patterns and food availability.

		Base year	Reference scenario 2050		Reduced human-edible feeds in ruminant rations		
Indicator	**Unit**	**Absolute**	**Absolute**	**Relative to base year**	**Absolute**	**Relative to base year**	**Relative to reference scenario**
Land occupation of arable land	million ha	1,354.6	1,541.4	*13.8%*	1,411.3	*4.2%*	*-8.4%*
Land occupation of permanent grasslands	million ha	3,378.6	3,378.6	*0.0%*	3,378.6	*0.0%*	*0.0%*
Land occupation agricultural land (total)	million ha	4,733.2	4,920.0	*3.9%*	4,790.0	*1.2%*	*-2.6%*
Energy supply for human nutrition	kcal/head*day	2,733.9	3,009.5	*10.1%*	3,170.3	*16.0%*	*5.3%*
Protein supply for human nutrition	g CP/head*day	72.2	82.4	*14.1%*	82.0	*13.5%*	*-0.5%*
Share of energy from animal products	% kcal	15.4%	18.1%	*17.1%*	14.4%	*-6%*	*-20.0%*
Share of energy from plant products	% kcal	84.6%	81.9%	*-3.1%*	85.6%	*1%*	*4.4%*
Share of protein from animal products	% CP	34.2%	39.4%	*15.3%*	32.7%	*-4%*	*-17.1%*
Share of protein from plant products	% CP	65.8%	60.6%	*-7.9%*	67.3%	*2%*	*11.1%*
Number of cattle	million animals	1,392.8	1,846.8	*32.6%*	1,264.9	*-9.2%*	*-31.5%*
Number of buffaloes	million animals	184.0	244.0	*32.6%*	238.7	*29.7%*	*-2.2%*
Number of goats	million animals	861.6	1,321.8	*53.4%*	1,244.4	*44.4%*	*-5.9%*
Number of sheep	million animals	1,097.4	1,683.5	*53.4%*	1,634.9	*49.0%*	*-2.9%*
Number of pigs	million animals	920.0	1,153.1	*25.3%*	1,159.9	*26.1%*	*0.6%*
Number of chickens	million animals	17,557.3	34,441.9	*96.2%*	34,895.9	*98.8%*	*1.3%*
Global warming potential	Gt CO2-eq	7.8	10.2	*32.2%*	9.0	*16.3%*	*-12.0%*
N-surplus	Gg N	116.5	173.5	*48.9%*	138.4	*18.8%*	*-20.2%*
P-surplus	kg P2O5	51.0	76.3	*49.6%*	65.3	*27.9%*	*-14.5%*
Pesticide use intensity	dimensionless	14.1	16.1	*14.1%*	15.2	*8.2%*	*-5.2%*
Soil erosion (water)	Mt of soil lost	33,706.8	36,772.1	*9.1%*	35,618.5	*5.7%*	*-3.1%*
Freshwater use	km3	1,135.4	1,277.0	*12.5%*	1,247.5	*9.9%*	*-2.3%*

4. Discussion

Our analysis shows for the example of dairy production that per kg of milk GWP would increase, while other environmental impacts will decrease (e.g. energy use, freshwater use, not presented in this paper), assuming a 20% decrease of global productivity in a scenario where human-edible feeds in ruminant diets are reduced to zero. Land occupation is an impact category which has a special importance for our study as we distinguish between arable land and permanent grassland. Permanent grassland covers about 2/3 of the utilized agricultural area globally (FAOSTAT, 2013). It is a resource with can hardly be utilized for human consumption without ruminants. Arable land on the other hand, is scarce and feed production on arable land directly competes with food production.

Using our model, we can take into account the absolute global scarcity of arable land and its indications for production and consumption patterns. We could demonstrate that there is a causal linkage between production patterns and dietary patterns, as the mere physical unavailability of products that are forecasted to be demanded increasingly in 2050 will not be present if concentrate use will decline. This takes into account the non-linearity of the impacts of the ruminant production system, i.e. once the existing grassland is occupied, arable land needs to be made available.

Attributional LCAs focus on strategies that enhance resource efficiency. The other sustainability strategies that concentrate on consistency and sufficiency (Schaltegger *et al.*, 2003) cannot be analyzed in a sound way with single-supply chain LCAs. Experts agree that pure efficiency gains may not be sufficient for global sustainable development in the food sector (Smith *et al.*, 2013). Therefore, fundamental changes in the agri-food sector should not be decided upon on a single-supply chain LCA basis only, but complemented with global impact assessments of long-term scenarios both using physical mass balance models as presented here and consequential LCAs, covering market interactions.

5. Conclusion

At global level, we face a trade-off between three strategies of sustainable development: efficiency, consistency and sufficiency strategies. Even if resource efficiency is maximized, we will global planetary boundaries are likely being overstepped. Sustainable intensification can help improving carbon and environmental footprints of livestock production. However, we have shown in this paper that depending on the perspective, LCAs may come to fundamentally different conclusions. In order to complement the efficiency strategy with other sustainability strategies dietary patterns need to be considered. Consistency strategies, e.g. organizing the food system closer to the natural cycles by feeding ruminants on grass, and sufficiency strategies, e.g. reducing meat demand, can complement efficiency strategies for fostering sustainable development at global level.

The model allows analyzing sustainability-related research questions with respect to resource efficiency, consistency and sufficiency. A formal functional unit (either protein or calories provided for human nutrition, can even be defined at global level. So, sustainability strategies should refer to both production and consumption and models for the assessment of such strategies should consider both production and consumption parameters as variables and not as constants. Our approach of a global physical mass balance model using LCI data is a tool complementary to single-product attributional and consequential LCAs for seeking sustainable solutions to global challenges.

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Comparison of CO_2e emissions associated with regional, heated and imported asparagus

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ABSTRACT

Carbon footprint of Asparagus, a major crop with 300,000 ha worldwide, was based on primary farm data. These included four Asparagus sources to extent the short cultivation period in Germany a) forcing local asparagus by soil heating, or sourcing of imports b) from e.g. Turkey or Greece by truck or by c) ship or d) plane from Peru, different supply chains and the unproductive first years after planting within 11 years of an asparagus plantation - according to PAS 2050 for horticulture (BSI, 2012), using 1 kg asparagus as functional unit with system boundaries from plantlet to end of life at consumer stage. Environmental-friendly cultivation was with waste heat viz energy from a local soot factory; therein, PE tubing, and pumps for the hot water dominated in the product carbon footprint (PCF) of 1.75 kg CO_2e/kg for the heated local asparagus. In case of the Peruvian asparagus, air-freight played the major role in comparison to transport by container ship. Regional asparagus scored best with 0.82 – 1.3 kg CO_2e/kg including 29 g/kg biogenic carbon as offset, whereas air-freighted Asparagus from Peru had 8 kg CO_2e/kg similar to 12.2 kg CO_2 e/ kg air-freighted asparagus by Stoessel et al. (2012).

Keywords: Asparagus, airfreight, carbon footprint, import, greenhouse gas

1. Introduction

Asparagus is chosen as a prime example for a carbon footprint analysis with different produce sourcing and marketing strategies. With ca. 24,000 ha, asparagus is not only the largest vegetable crop in Germany, but a major vegetable with 300,000 ha worldwide with increasing popularity in a healthy diet. Expansion of the short cultivation period between May and June in Germany, however, can be achieved only by a) forcing asparagus by soil heating, sourcing by imports b) from e.g. Turkey or Greece by truck or by c) ship or d) plane from Peru. The objective of this contribution was to evaluate the carbon footprint of these four asparagus sources to aid consumer decision-making and determine GHG hot spots and climate-friendly sourcing.

The challenge is to combine the largest field grown vegetable commodity in horticulture with the new PAS 2050-1 for horticulture. The only two carbon footprint studies on asparagus were in the UK (Audsley et al., 2009) and Switzerland (Stoessel, 2012) before the PAS 2050-1 hort standard (BSI, 2012) became available. For a number of reasons, neither biogenic carbon nor the relevant use phase was included in these previous studies. For the first time, the new PAS 2050-1 (hort) is used as methodology including the new approach to biogenic carbon and Land Use Change (LUC) emissions for the shorter time period of only 20 years (old 100 years) and different crop rotation systems. This study was part of the pilot projects towards the development of the PAS 2050-1 hort (BSI, 2012).

Hence, the objective of this study is to assess the carbon emissions of different asparagus field production systems using a number of supposedly improvements such as waste energy for forcing asparagus to satisfy consumer demands for all year around vegetable supply. Since regional asparagus is not available all year round, the airfreight import of fresh asparagus from other parts of the world is used to fill in the marketing gap. Asparagus is employed here as a model crop for the comparison of heated and un-heated early (forcing) local production. Import of asparagus from Peru by ship and airfreight is included as a different way of sourcing for the European market.

2. Methods

2.1. Primary data assessment

Primary data from two farms in Germany with different supply chains include the unproductive first years after planting within the overall 11 years of an asparagus plantation. Biogenic carbon was also accounted for following the special requirements of the new PAS 2050-1 for horticulture (BSI, 2012). A heated cultivation was compared with an unheated standard cultivation system and marketing via two different ways (retail and farm shop).

2.2. Calculation of carbon footprint

The carbon footprint was calculated for the supply of asparagus based on the new PAS 2050-1 "Assessment of life cycle greenhouse gas emissions from horticultural products - Supplementary requirements for the cradle to gate stages of GHG assessments of horticultural products undertaken in accordance with PAS 2050" (BSI, March 2012), which includes the GHG emissions CO_2 (1x), CH_4 (25x) and N_2O (298x) and results in CO_2 equivalents (CO_2e). The new horticultural PAS 2050-1 (BSI, March 2012) includes specific requirements for different cultivation systems like perennial crops such as asparagus or orchards over their complete lifetime from plantlet supply to grubbing. Additionally, there is the opportunity to include the biogenic carbon in the harvested product, if used for food and feed, which is not considered in previous studies.

2.3. Functional unit and system boundary

Two functional units were employed for carbon footprint assessment:

- Acreage [ha] for asparagus cultivation (PAS 2050-1)
- Weight [kg] of saleable product at farm gate (PAS 2050-1) and for further marketing, use phase and disposal (PAS 2050)

The results for the assessed greenhouse gas emissions (GHG) are shown for the cultivation phase up to the harvest with the specific Farm Carbon Footprint (FCF), including the post-harvest activities up to the farm gate (B2B) and with transportation, marketing, consumer use phase and disposal (B2C, PCF).

2.4. Cultivation and marketing systems for asparagus

Three cultivation systems on two farms were selected on the grounds of representativeness, different primary energy use and marketing strategies. The large farm used one cultivation system with an underground soil heating system and another one without heating system. On the second farm the overall asparagus acreage was smaller and not heated. The marketing strategies differed in that the large enterprise (farm one) with its large volume of asparagus markets exclusively via nationwide wholesale and retailers. By contrast, farm two sales the smaller volumes of asparagus through farm shops and regional street markets direct to the consumer. Consumer shopping tours with private vehicle differ in distance and size of the shopping basket (20 kg with 1 kg asparagus for retail and various scenarios depending on specific marketing system combined for farm two). Home storage and cooking as well as composting were modeled.

3. Results

3.1. Carbon Footprint of Asparagus

The carbon footprint of asparagus was calculated over the entire life-span including the productive and unproductive phases, i.e. 11 years. In this case study for the new PAS 2050-1 only one farm with one cultivation system was analysed to study the ease of implementation of the new rules. LUC was calculated with an Excel tool that was developed in the pilot project. The result of the LUC for asparagus following annual crop land was negative for our specific growing and environmental conditions. To avoid any criticism of offset it was assumed zero in our calculation. The biogenic carbon was calculated from its 6 % dry matter with 47 % carbon content based on average (11 year) yield (7t/ha) including the unproductive phase of the cultivation. The result for the Farm Carbon Footprint (FCF) was 2.307 t CO_2-eq/ha asparagus, after the biogenic carbon was subtracted.

Including the cooling, grading and packaging the business to business carbon footprint according to the new PAS 2050-1 was 414 g CO_2-eq/ kg asparagus (Table).

The transportation to retail and the use phase (shopping tour, fridge and cooking) at the consumer amounted to 401 g $CO_{2\text{-eq}}$/ kg asparagus using the guideline of the PAS 2050:2011. The overall business to consumer (B 2 C) carbon footprint of 815 g $CO_{2\text{-eq}}$/ kg asparagus shows the result over the all life-cycle stages of asparagus.

Since waste energy was employed from a separate enterprise, viz a soot factory, this energy was excluded from the asparagus carbon footprint. However, the carbon footprint of heated asparagus doubled to 1.75 kg CO_2e/kg relative to unheated asparagus due to the plastic tubing to distribute the hot water to the plants and due to the energy consumption to pump the hot water

Table 1: Carbon footprint of asparagus cultivated in an integrated production system (asparagus regional)

	Integrated Production (IP) of Asparagus
Farm carbon footprint per acreage [t/ha]	
Average asparagus yield per ha and per year (over 11 years)	7.02 t/ ha
Yearly cultivation, tillage and planting (per ha) without bio-genic carbon	2.503 t CO_2-eq/ ha/a
LUC (asparagus after annual cropland)	0.0 kg CO_2-eq/ ha/a*
Biogenic carbon per ha (6 % dry matter of asparagus)	-0.198 t CO_2-eq/ ha/a
From planting to harvest (over 11 years) (ha)	2.307 t CO_2-eq/ ha/a
Farm carbon footprint [g per kg]	
From planting to harvest (over 11 years) (kg)	328 g CO_2-eq/ kg asparagus
Cooling, grading and packaging (5 kg cardboard carton)	086 g CO_2-eq/ kg asparagus
Carbon footprint from seedling to harvest according to PAS 2050 -1 (B 2 B)	**414 g CO_2-eq/ kg asparagus**
Product carbon footprint [g per kg]	
Overall transportation farm to retail	096 g CO_2-eq/ kg asparagus
Use phase (shopping tour, fridge and cooking)	305 g CO_2-eq/ kg asparagus
Product Carbon Footprint from harvest via use phase to disposal	**401 g CO_2-eq/ kg asparagus**
Product Carbon Footprint (PCF) CO_2 eq /kg asparagus **(B 2 C)**	**815 g CO_2-eq/ kg asparagus**
Product Carbon Footprint (PCF) CO_2 eq /kg asparagus **(B 2 C) excl. offset of biogenic carbon**	**843 g CO_2-eq/ kg asparagus**

*LUC result was negative but assumed as zero.

Alternatives to heated regional asparagus in spring for the consumer are imports by truck from Greece (2,100 km by ship or plane from Peru, resulting in 6- 12 kg CO_2e/kg asparagus in comparison with forced regional asparagus 1-2 kg CO_2e/kg asparagus.

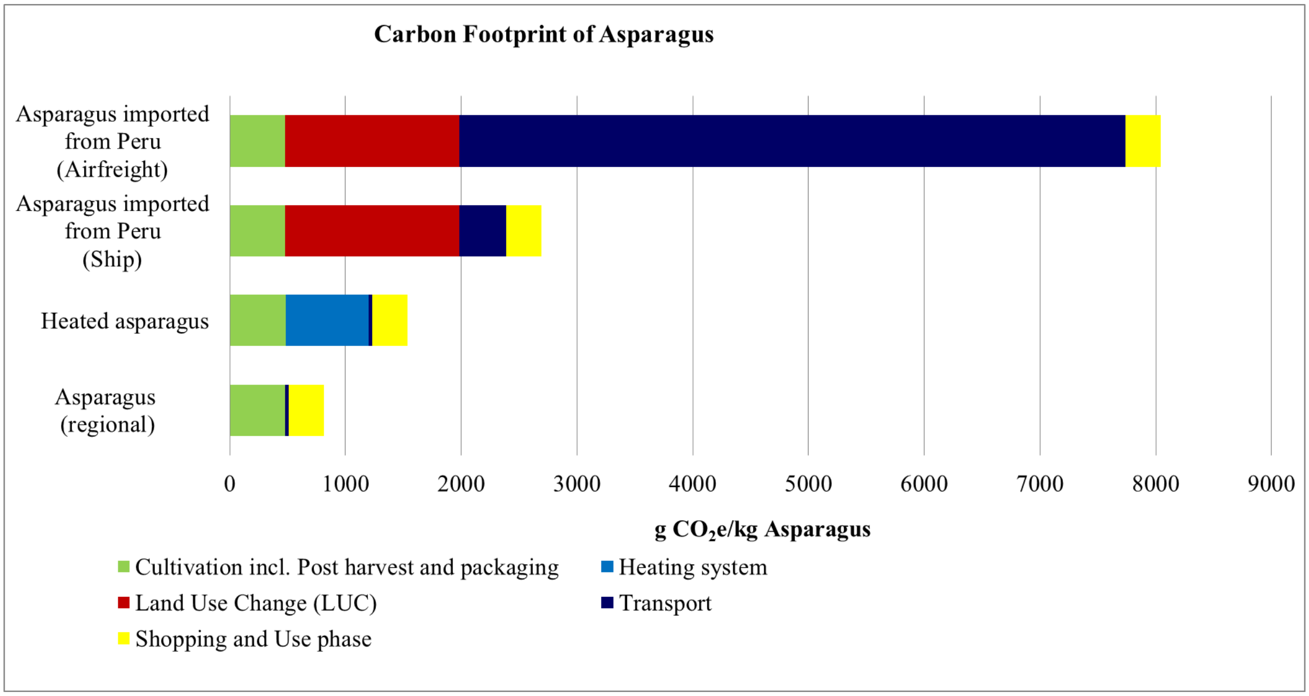

Figure 1: Product Carbon Footprint of different sources of Asparagus

3.2. Biogenic carbon

Biogenic carbon, i.e. the carbon contained in the harvested product exported outside the system boundary (out of the farm gate), is offset in the PAS 2050-1 against the carbon emissions during horticultural production, if used for food or feed. Similarly, bio-genic carbon not used for food or feed, e.g. tree trunks for the timber or furniture industry, can be offset against horticultural production (BSI, 2012) (Table).

Capital goods in horticulture such as greenhouse support structures for poly-tunnels, buildings, grading facilities and cold stores are defined and excluded from the Carbon Footprint calculations in PAS 2050-1.

Consumables, which are replaced on a regular basis, such as plastic foils for plant covering and growing substrates, are included. Similarly, fertilisers and plant protection compounds are included in the product carbon footprint.

4. Discussion

The larger carbon footprint of 1.9-2.4 kg CO_2e/kg asparagus by Audsley et al (2009) for asparagus cultivated in England may be due to the calculation from basic root vegetables and adjustments made for yield deviations from the average root vegetable crop. Our value of 8 kg CO_2e/kg airfreighted asparagus from Peru compares favorably with 12.2 kg CO_2 e/ kg asparagus by Stoessel et al. (2012), who used a stopover on the flight to Switzerland compare with a direct flight to Frankfurt in the present study.

5. Conclusion

The present study is, to our knowledge, the first approach based on the new dedicated PAS 2015-1 (2012) and utilizes comprehensive primary data assessed on carefully selected farms. Different marketing scenarios are presented, which include farm sale (regional asparagus) and supermarket shopping (Peruvian asparagus), in which the consumer played a relevant role with his shopping trip (2 x 5 km) with a standard 20 kg shopping basket with 1 kg asparagus (Schaefer, 2012).

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Democratization of Food Environmental Product Declarations, with a Beer Example

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Abstract
Type III EPDs are becoming more and more common, and are a must-have in some fields. However, the cost of developing them is high, leaving small companies at an extreme disadvantage. To address this economic barrier, IERE has developed automated EPDs using beer as an example. The system depends on a deep understanding of the process and the range of options for making the product, but the results is a piece of software that can be used by any brewer with only business information needed as inputs to the calculation. Calculations are in real time, feedback as to sources of impacts is given to the brewer, and the public can see the results in an online EPD. The cost is reasonable, with the lowest fees only $150 per year, allowing EPDs to be created for each batch of beer, providing ISO compliant EPDs for a few dollars each. This model can be applied to all food products, with the outcome of a democratized EPD system.

1. Introduction

Type III Environmental Product Declarations (EPD) are an increasingly common tool used by companies to communicate the environmental impacts of their products. They are especially useful for business to business (B2B) communication as organizations attempt to better understand the impacts of their supply chains. The use of EPDs is also being driven by industry groups such as the US Green Building Council (USGBC) and even government as is the case with the French Grenelle Environnement.

Generating EPDs, performing the life cycle assessments (LCA), and undergoing third party verification can be costly endeavors. Some companies, especially small and medium enterprises (SMEs) such as farmers and small food processors, may find these costs prohibitive and forego EPDs thereby putting them at a competitive disadvantage.

In an attempt to address the cost barrier to EPD adoption several programs, including those mentioned above, have made allowances for the use of industry average or sectorial EPDs within a product category.

However, these average EPDs inherently fail to meet many, if not all, of the objectives of Type III environmental labels as defined by ISO 14025:2006:

a) "to provide LCA-based information and additional information on the environmental aspects of products";
b) "to assist purchasers and users to make informed comparisons between products";
c) "to encourage improvement of environmental performance";
d) "to provide information for assessing the environmental impacts of products over their life cycle."

While industry average EPDs may provide some value with respect to objectives "a" and "d" they lack the ability to differentiate similar products or incentivize environmental improvement. This paper proposes an alternative approach to increasing the use of EPDs while keeping costs low by using an automated EPD system. A pilot study that utilized an automated system to generate EPDs for multiple products within the same product category was conducted with micro-breweries in the Pacific Northwest.

2. Method

2.1. Automation Overview

An automated EPD system is not only cost effective, it also mitigates many of the issues associated with comparability among EPDs by providing a consistent modeling framework and background data. The process PCR development addresses many important modeling decisions and assumptions (system boundary, functional unit, cut-off rules, data quality requirements, impact assessment results, and reporting requirements). However, there are typically several instances where decisions are left to the discretion of the LCA practitioner. The result is potentially large variation in model design and impact results for EPDs that are published under the same PCR.

An automated system takes EPD standardization a step further than the PCR by using a single modelling framework, default background data, default assumptions, and a simple user interface that is accessible by non-technical users.

The pilot study developed an automated EPD system for a group of small-scale brewers that would not have the time or money to develop a custom EPD for one of their products, let alone all of the different beers they produced. The tool is available on an annual subscription basis and subscribers are able to generate EPDs for as many batches as they would like. Users can perform what-if analyses when planning new brews or to investigate opportunities for reducing their impacts.

2.2. System Design

The LCA model used in the EPD tool conforms to the Earthsure Beer Product Category Rule, number 50202201:2012 (IERE, 2012) as well as ISO 14040, 14044 and 14025 (ISO, 2006; 2006b; 2006c). The functional unit for the EPD is twelve ounces of beer delivered to the point of consumption. However, the tool allows users to calculate impacts associated with other commonly used units (e.g. 16 oz., ½ barrel). Figure 1 shows the system boundary used for the EPD system's underlying LCA model. The beer life cycle is separated into three life cycle stages. Upstream production includes the raw material use, manufacturing and transport of all inputs to the brewery stage. Beer production includes all brew house operations from grain mashing, through boiling and fermentation, to packaging and cleaning of equipment for the next batch. The use phase consists of distribution to the point of consumption (either retail or restaurant) as well as treatment of waste products. Capital goods, employee activity, and recycling of discarded materials are excluded from the study.

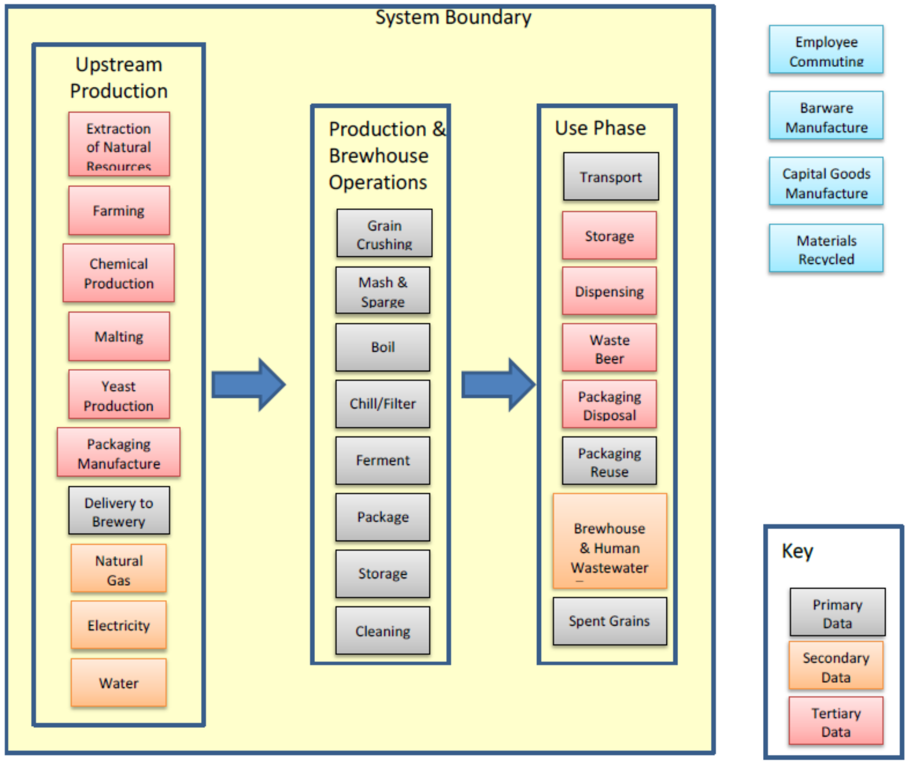
Primary data was collected for all unit processes under the direct control of the brewery with a reference flow of one batch. Information on the beer recipe, production volume, equipment usage and packaging were input directly by the brewers into the software.

Production data for agricultural products was obtained from USDA National Agricultural Statistics Service Information, USDA National Agricultural Library Digital Commons (USDA, 2012), and the Ecoinvent database[vi]. Material content data for cleaners and other chemicals were obtained from Material Safety Data Sheets (MSDS) provided by suppliers and linked to life cycle inventory data from the Ecoinvent database.

Regionally specific process LCI data was used whenever possible. This included the creation of an electrical grid mix to represent each brewery's local electric utility. When regional process data were not available, US average process LCI data were used.

The cutoff rules used specified that at least 95% of all mass and energy and all known toxic materials are included in the analysis. No product representing more than 1% of the total mass or energy of the system was excluded. Life cycle impact assessment was performed using the US EPA TRACI 2.1 methodology (Bare, et al, 2002). Total life cycle inventory amounts for water use and land occupation are also calculated per the PCR guidelines.

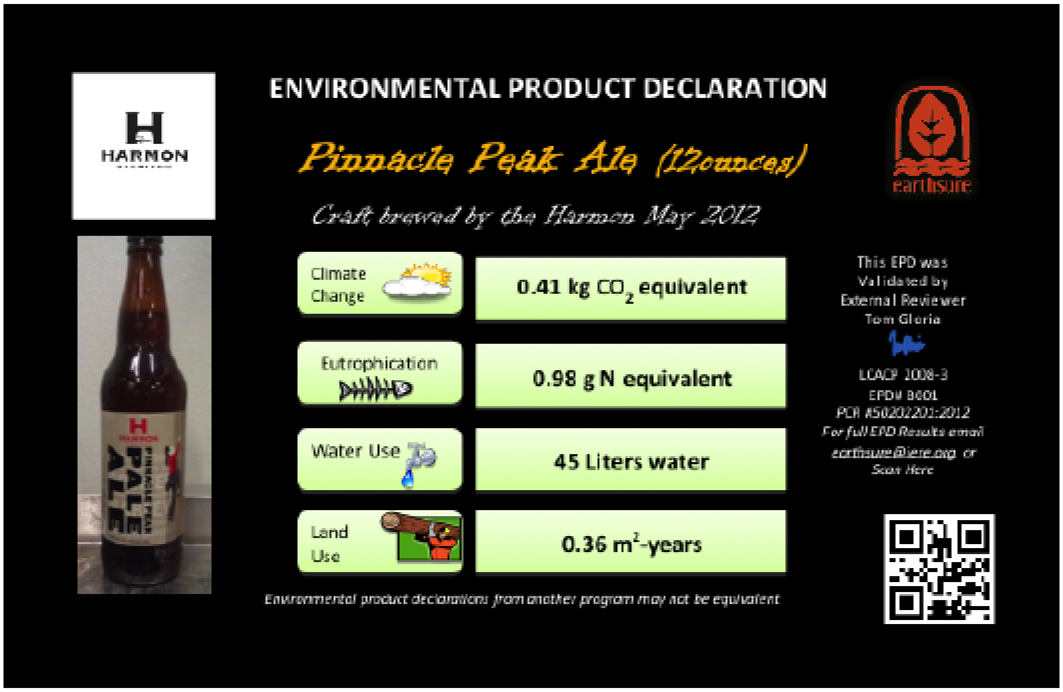
Figure 1: LCA System Boundary



3. Results and Discussion

Figure 2 shows an example EPD created by a brewer that participated in the pilot. While the reported impacts for each beer analyzed were generally of the same order of magnitude, the largest sources of those impacts varied for several of the brewers. The environmental impacts of the beer life cycle are primarily related to material inputs, brewery operations, and beer consumption. The ingredient phase of the life cycle is the main contributor to ecotoxicity, ozone depletion, and water use at all five breweries; it is also the prime factor in eutrophication, land use, and smog at four breweries. The brewery phase is the primary contributor to climate change and acidification at three breweries; this phase has the highest smog value at a single brewery. The consumer use phase contributes heavily to climate change given the release of CO_2 when beer is consumed. The packaging phase is not a common prime source of impacts among the breweries; nonetheless, it is the main contributor to eutrophication and land use at an individual brewery.

Figure 2: Example Beer EPD



The EPDs were made publically available online for customers to view and compare the impacts of the different beers. In addition, the brewers were able to use the results in their own marketing materials. The pilot was able to achieve the goal of producing a high quality EPD for a group of small and medium enterprises at an affordable price. Cost per EPD varied based on the size of the brewery and the number of EPDs each generated. However, the costs were still quite low, especially compared to conventional EPD development, with one brewer paying the equivalent of $50 per EPD.

The pilot also showed that EPDs can be an effective tool for companies to measure their environmental impacts and provide incentive to improve the environmental performance of their production. All five of the participating brewers have modified or plan to modify their practices after creating their EPDs. Examples of these changes include: insulating storage tanks to reduce energy requirements; changing cleaning procedures to limit chemical use; investing in energy efficient equipment; the purchase of an electric vehicle for distribution.

4. Conclusion

The pilot study showed that an automated system is capable of providing accurate, verifiable and cost effective EPDs that adhere to the goals and ISO 14025. This was achieved at a low cost per EPD and required little technical knowledge on the part of the EPD owner. More importantly, it provided companies that are interested in sustainability with valuable information that they can now use to reduce their environmental impact. The brewers would likely not have access to this information otherwise as they do not have the time, money or expertise required to generate EPDs in a conventional manner.

Such a system bridges the gap between a one-time single product LCA and a sector level average by utilizing primary data for operations occurring within the organization. Because these operations are under the direct control of the organization, they are most likely to be the source of differences between similar products. The use of default data for all other background processes significantly reduces the resources required for data collection when compared with a traditionally developed EPD.

Many more such automation systems are needed to bring EPDs into the hands of the small producer and subsequently their customers. Agricultural products are prime candidates for EPD automation due to the similarities in the production of a crops as well as the large number of farms. We estimate that crop EPDs could be produced at an even lower price point than the beer pilot, further democratizing EPDs. Information technology coupled with parameterized LCA models can be a force for achieving better environmental performance for food systems.

5. Acknowledgements

The pilot EPD automation study was supported by the American Brewing Co., Fort George Brewery, Harmon Brewing Company, Hop Works Brewery, The RAM Restaurant and Brewery, Vashon Brewery, and the City of Tacoma Sustainability Program.

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EcodEX: A simplified ecodesign tool to improve the environmental performance of product development in the food industry

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ABSTRACT

Nestlé has engaged in the development of a simplified ecodesign tool (EcodEX) for the assessment of products in the development process. EcodEX guides the user along the predefined phases of the life cycle of a typical food product, using a preselected list of life cycle inventory data. The tool is currently being used by a user base of non-LCA-specialists that have been provided with a basic training on the tool use, as well as with basic understanding of sustainability in the food sector. First results indicate that the tool can improve the design of new products and plays a very important role in increasing the understanding of sustainability in the organization. Key challenges are the availability of life cycle inventory data and the training and support process for a large user base.

Keywords: product development, innovation process, ecodesign, simplified LCA,

1. Introduction

Nestlé, the world's leading Nutrition, Health and Wellness Company, has integrated the life cycle approach into its innovation and renovation processes to improve the environmental performance of new food products under development. Over the past years, Nestlé has done life cycle assessments (LCAs) for representative products in each of its business categories, and has summarized and shared them internally using Sustainability Category Profiles. Some of these studies have also been published (Humbert et al, 2009a; Humbert et al, 2009b). While these studies give valuable insight for strategy setting of businesses, the conventional life cycle assessment approach is impractical for widespread application in product development. A company as large and diverse as Nestlé, with thousands of new product developments every year, would have to deploy a considerable amount of resources on these assessments. Furthermore, given the speed at which innovation occurs in the fast moving consumer goods sector, and in particular in the food industry, results from conventional LCA studies will, in many cases, become available only once the Nestlé Product Development Process has advanced significantly (see figure 1 below). Therefore, a simplified, yet reliable, ecodesign tool specific for the assessment of food and beverage products has been developed and rolled-out in Nestlé R&D over the last years. This paper presents the outcomes of this roll-out and assesses the benefits of such an approach for large food companies.

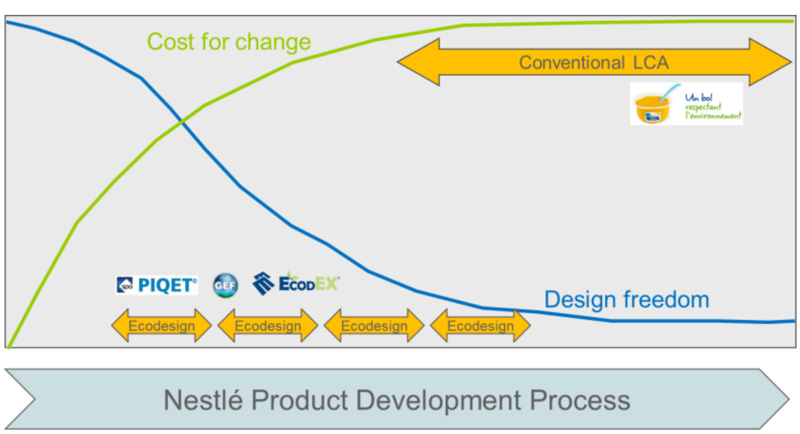

Figure 1. Nestlé Product Development Process and the positioning of conventional LCA (top right) vs. simplified ecodesign tools, which apply the LCA methodology earlier in the design process to optimize their influence on product development.

2. Methods

2.1. Goal and scope definition

EcodEX is a simplified ecodesign tool that guides the non-expert user through the different stages of the life cycle of a typical food product, providing him/her with a reliable estimate of the environmental hotspots in the products value chain, as well as a comparison of the environmental performance of different design alternatives (Selerant 2014). As compared to conventional LCA software, EcodEX has a significantly reduced flexibility in what can be modelled with the tool: the user is guided across the typical life cycle of a food product (ingredients, packaging, processing, distribution, consumer use, and end of life) and can enter only the data appropriate for these life cycle phases. The user mostly selects LCI (life cycle inventory) profiles from a pre-defined list without the possibility of modifying them. In the case of processing, though, default values for electricity, heat, and water consumption may be overwritten with measured factory data.

The tool is linked to the Nestlé recipe management system, extracting high quality data on recipe composition and processing steps directly from an existing IT system. Ingredient specifications in the food industry can easily run into tens of thousands, turning their management into a complex endeavor. This is why a key requirement in the development of the tool was the link to the management system already in place at Nestlé.

2.2. Inventory data

The recipe management data and information entered by the user (e.g. mode of transportation and distance travelled) are then combined with inventory data. The majority of LCI data comes from public data sources, in order to ensure the credibility of the results of the tool. Major databases used in EcodEX are Ecoinvent 2.2 (Frischknecht et al, 2005) (for electricity profiles, transportation, as well as packaging materials), and the World Food LCA Database (Quantis, 2014) (for food ingredients). Also, the Agribalyse database (ADEME, 2014) may be used in the future for additional food ingredients. Some ingredients where no datasets are currently available in public databases have been created based on LCA studies done by consultants for Nestlé. Datasets on novel food ingredients or packaging materials may also come from suppliers.

2.3. Impact assessment methods

The tool calculates environmental impacts according to internationally harmonized LCA methods, such as the ISO 14040 & 14044 norms (ISO 2006a, ISO 2006b), the European Union Product Environmental Footprint (PEF) (EC 2012), and the EU Food Sustainable Consumption and Production Roundtable ENVIFOOD protocol (Food SCP RT 2013). Currently, five environmental impact indicators are taken into account by EcodEX: land occupation and water consumption at the inventory level; GHG emissions at a 100 year perspective (IPCC 2007) and Non-renewable minerals and fuels (CML 2014) at the midpoint level; and Ecosystems Quality (based on the IMPACT 2002+ method and modified to exclude land occupation and thus avoid double counting) at the end-point level (Jolliet et al, 2003).

In addition to the exclusion of land occupation in Ecosystems Quality, the ecotoxicity characterization factors of metal emissions in agriculture have been set to zero. This is because the modeling of metal toxicity in IMPACT2002+ is deemed insufficiently accurate to be taken into account in a simplified ecodesign tool for the agri-food sector. For instance, the impacts of copper used in coffee and cocoa production (application rates of copper pesticides are well below the levels authorized per hectare in organic agriculture for many crops in Europe) result in toxic effects several orders of magnitude above those of “problematic” conventional pesticides such as Aldicarb that are also used in coffee and cocoa production.

2.4. Interpretation

A consequence of the development and application of a simplified ecodesign tool is the introduction of larger uncertainty in the results of the assessments. Therefore, EcodEX is not intended for communication purposes, but for internal decision making only. Work is currently ongoing to develop a specific module to assess uncertainties in a reliable way directly as part of and EcodEX assessment.

3. Results

EcodEX has been regularly used by the Nestlé R&D community for approximately one year now. There are around 100 users of the tool, and approximately 900 LCA scenarios have been calculated in this period. The results of these scenarios have influenced the development process of new products in several ways:

If the environmental performance of the new design worsens:

- Product designers may work on an alternative design that improves the environmental performance of the product. In the past, in the absence of EcodEX, designers were not aware that their designs had worsening impacts for the environment.
- The product development may be interrupted or stopped altogether because environmental performance is of key importance to the business unit and market. Without the information provided by EcodEX, decision makers are not in the position to take timely decisions at the earlier stages of the development process. In case an LCA study was commissioned to an external consultant, the conclusions would become available too late in the design process.
- A decision may be taken where the project goes ahead despite the adverse environmental effects, because of other criteria such as consumer preference, nutrition, quality or legal requirements. In this case, EcodEX did not have any tangible consequence for the environment. However, decision makers will be aware that they are launching a product with higher environmental impacts, and will need to be able to defend their decision e.g. against critical consumers or NGOs.

If the environmental performance of the new design improves, the results can be shared internally to gain support for the new product development. This may result in a faster adoption of the product, or the use of the product in other markets.

In addition to these direct benefits on the product development, the tool has also very strongly increased the understanding on food environmental performance among the users: product developers that calculate the environmental performance themselves get a much deeper understanding of their product life cycle than those who are presented with a report and finalized results. For instance, everybody knows intuitively that meat products have higher environmental impacts than plant-based alternatives. However, if the product developer is able to calculate the difference himself, the actions prompted will be larger and further reaching because the product developer fully engages with the assessment.

4. Discussion

A number of challenges have occurred during the development and implementation of EcodEX:

4.1. Inventory data

For such a tool to be applicable, high quality LCI data for an extensive array of food ingredients used in the food industry are required. This is currently being addressed by a number of initiatives, such as the World Food LCA Database Project (reference). Currently, LCI profiles are available for the majority of key ingredients, but too specific profiles are often not representative of general conditions (e.g. tomato, organic production, Switzerland vs. tomato, global average). First of all, ingredient data representing global averages should be developed. Over time, the amount of data will increase, and will allow the assessment of more specific ingredients (e.g. stevia vs. artificial sweetener), ingredients from specific regions grown under specific climatic conditions (e.g. wheat from Southern Spain vs. wheat from the Nebraska, US), and different cultivation practices (e.g. integrated pest control vs. conventional production).

4.2. Training

Users have to be extensively trained to gain the appropriate understanding on sustainability and life cycle assessment, which inevitably requires time and significant resources. Most users have a background in nutrition, food science, process engineering, or packaging technology, which provides them valuable insight in their focus area. In order to be able to make meaningful decisions on the environmental impacts of the products they develop, they need to expand their skills set to include this topic. It cannot be expected that the quality of the assessments is comparable to that of a trained LCA specialist at the beginning. Practice and frequent use of the tool, along with ongoing training through webinars (conference calls with desktop-sharing) and a review of the studies performed will assure that EcodEX users make high quality assessments over time.

4.3. Alignment with external initiatives

The importance of life cycle assessment for external communication is likely to increase in the future. While EcodEX is intended for internal decision making, EcodEX users will want to use the results for external communication as well. Therefore, it is important that the methodology in the tool is as closely aligned with externally developed methodologies as possible. This is not always a straightforward task, because external methodologies are subject to change and not all requirements are known in sufficient details.

5. Conclusion

Our experience shows that the use of simplified ecodesign tools can be very useful tool to improve the environmental performance of the products of a food company during their innovation and renovation process. In the future, the LCI data in the tool will be complemented with new profiles to make more specific assessments possible. The functionality of the tool will also be extended, in particular to accommodate a module that assesses uncertainty and significance of differences. This will further increase the usability of EcodEX and help improve decision making.

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Life cycle assessment of the global food consumption

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ABSTRACT
Food consumption is a major driver of global environmental impacts. This paper presents an analysis of the life cycle impacts caused by global food consumption based on the newly completed input-output model; Exiobase v2. Exiobase v2 is a multi-regional input-output database covering 43 countries plus five rest-of-world regions for 2007. The data in the model includes all global product, emission and waste flows related to food consumption, i.e. a global mass flow analysis of all food related flows. The functional unit of the study was the global consumption of food in 2007. The included life cycle stages were cultivation/husbandry, processing, retail, preparation in households/restaurants and food waste disposal. The impact assessment focused on GHG-emissions and land-use. In this respect it should be noted that a model of indirect land-use changes is integrated in the Exiobase v2 model used in this study, to account for GHG-emissions caused by the use of land.

Keywords: Global food consumption, Input-output LCA, GHG-emissions, Land-use, Indirect land-use changes

1. Introduction

Food consumption is a major driver of global environmental impacts. Some studies give an indication of the overall magnitude of the global impact of food, e.g. sector specific LCAs (e.g. FAO 2006) and country/region specific input-output LCAs (e.g. Weidema et al. 2005). But yet, no studies have analyzed the total global life cycle environmental impacts from food consumption in detail.

This paper presents an analysis of the life cycle impacts caused by global food consumption based on the newly completed input-output model; Exiobase v2, which was created as part of the EU FP7 project CREEA (http://creea.eu/). Exiobase v2 is a multi-regional input-output database covering 43 countries plus five rest-of-world regions for 2007. Compared to other input-output models, the Exiobase v2 database also contains product flows in physical unit (mass and energy), and it is based on detailed mass balances for each product and industry, and it includes calculated quantities of food waste.

The functional unit of the study is the world's consumption of food in 2007. The included life cycle stages are cultivation/husbandry, processing, retail, preparation in households/restaurants and food waste disposal. The impact assessment focusses on GHG-emissions and land-use. In this respect it should be noted that a model of indirect land-use changes is integrated in the Exiobase v2 model used in this study, to account for GHG-emissions caused by the use of land. The results of the study are presented at the global scale, and they are broken down showing contribution analysis per food category, per life cycle stage, per sector, and per capita impact.

2. Methods and data

2.1. The Exiobase v2 input-output LCA model and mass flow analysis

The basis of the LCA is a detailed mass flow analysis of all flows related to the world's food consumption. This mass flow analysis is carried out using the approach developed in the EU FP6 project FORWAST (http://forwast.brgm.fr/) and extensively implemented at the global scale in high level of detail in the EU FP7 project CREEA (http://creea.eu/). The approach integrates mass flow analysis with economic input-output models, and the outcome of this is a hybrid input-output database where all flows with a physical mass are accounted in dry matter mass unit, energy flows are accounted in energy unit, and the remaining flows are accounted in economic unit. This database, the Exiobase v2, is very similar to a traditional process life cycle inventory database, but a few things differ:

- The level of completeness is higher; Exiobase v2 includes all the transactions in the economy and their associated emissions, whereas e.g. the ecoinvent database generally does not include activities in the service sector (marketing, accounting, legal services, cleaning etc.) and the sum of included activities in specific sectors may not add up to all different productions in this sector. This is also the reason why process based LCAs generally show lower results than input-output based LCAs.

- The level of detail is lower; the Exiobase includes 162 activities whereas e.g. the ecoinvent database includes thousands of activities/processes.
- The database includes transactions in actual flow volumes (e.g. the reference flow of wheat cultivation is equal to the global wheat production) as opposed to process databases, which are typically not integrating this information (they only include normalized processes; the wheat process only show transaction per unit of output of wheat).

The features of input-output databases make them ideal for macro-level analysis with little focus on the very detailed technology factors. But when it comes to life cycle inventories of more specific products (that differ from broader product categories in the input-output model), additional information from process based inventory data are needed.

The Exiobase v2 database is based on very detailed mass balances for each input to an activity in economy. The concept is illustrated in Figure 1. Since the mass balance principle illustrated in Figure 1 is applied to every input to activities/industry sectors in the model, the outcome is a fully mass balanced mass flow account of each included country and region. An example of the application of the principle in Figure 1 is: Raw milk production is an activity in the model. This activity has inputs of feed concentrate (product) and grass (resource extracted on the farm). These inputs become outputs of: milk and meat (supply of products), respiratory emissions (CO_2, CH_4 and H_2O), and manure (material for treatment). Further, for the respiration inputs of oxygen are also needed (resource inputs).

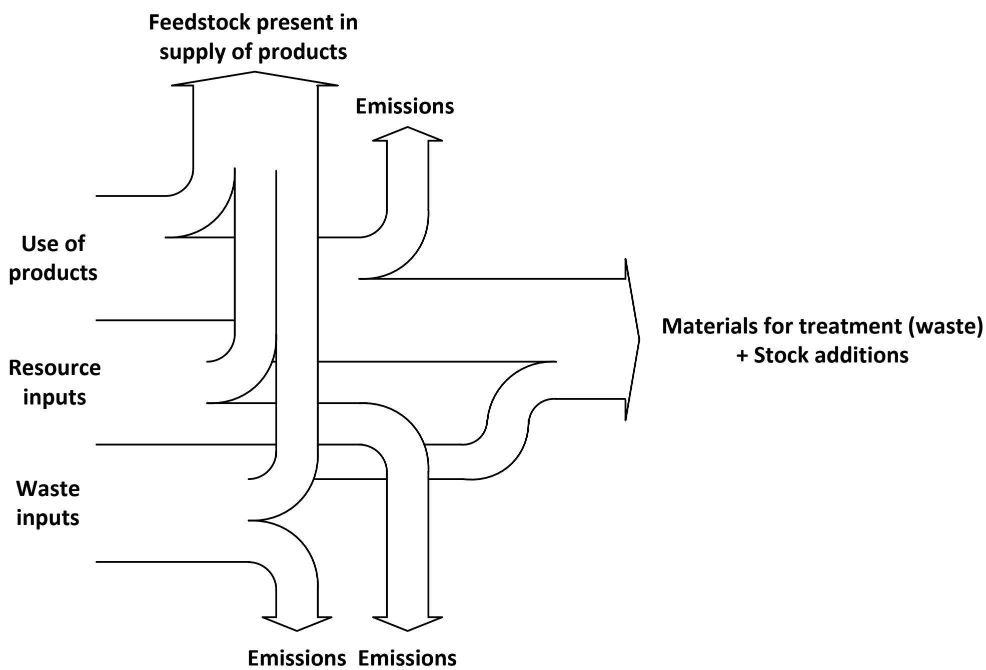

Figure 1. Mass balance concept in the model behind the Exiobase v2 database (Schmidt et al. 2012a). On the left side are the types of any inputs to any activities. The arrows represent the possible different fates of the inputs. Basically, an input (product, resource, waste) can become a product, an emission, a material for treatment (waste) or stock addition, i.e. a material that will become waste at a later stage (this could be an input of a tractor in year 2007 that will become waste 7-10 years later).

The level of detail of the model is 162 products/industries of which 13 are agricultural production of foodstuffs, 1 represents fishery, 11 are food productions, and 6 are treatments of food waste (e.g. landfill, incineration, and sewage treatment). The specific level of detail for food in the model can be seen in Table 1.

In terms of life cycle inventory modelling, by-product allocation is avoided by substitution in accordance with the ISO 14040 and 14044 standards on LCA (ISO 2006a,b). In input-output terms, substitution is equivalent to using the by-product technology assumption. This approach maintains all mass, energy, and economic balances which are also needed for the mass flow analysis (Suh et al. 2010). If allocation had been applied, it would not have been possible to present a mass-balanced product system as in Figure 2 (see e.g. Weidema and Schmidt 2010).

The data related to the food product chain used for the Exiobase v2 model includes data on the total production of crops, animals, fertiliser, and food products. These data are obtained from global statistics such as FAOSTAT, International Fertiliser Association, etc. For each agricultural and food product, a detailed mass balance was established so that inputs equal outputs. The flows of raw materials, energy and emissions from the involved industries are calculated based on e.g. LCI coefficients from existing life cycle inventories, calculated feed requirements (IPCC 2006), parameterized emission inventories based on IPCC (2006) and FAOSTAT (2013) while at the same time respecting national and global product balances, i.e. that the total supply of products equals the total use of products. Also the biomass production and animal and human metabolic balances are taken into account. However, for the biomass production, only the flows relating to photosynthesis are accounted for, i.e. only C, O and H are accounted for, while minerals like nitrogen, phosphorus and potassium in the products are not accounted for.

The used version of the Exiobase v2 database is a preliminary version. It has been aggregated to one global database by adding all 43 countries and five rest-of-world regions. Therefore, when the final version of the database is published, the results may change due to revisions.

2.2. Integration of indirect land use changes in the LCA model

In addition to the emissions accounted for in the Exiobase v2 model, the contribution from indirect land-use changes is also included. According to Le Quéré et al. (2012), around 9% of global CO_2-emissions originates from land-use changes. Therefore, it would be a significant underestimation of the real impacts of food if the land-use change induced emissions were not addressed. The concept of the applied iLUC model is described in Schmidt et al. (2012b and 2014) and the integration of the iLUC model in the input-output framework and the quantification of the iLUC model are described in Schmidt and Muñoz (2014, chapters 3 and 5).

2.3. Using the Exiobase v2 input-output model for LCA of the world's food consumption

When ascribing the flows in the Exiobase v2 database to the functional unit, all food for consumption in the input-output table needs to be identified. Food is generally consumed in three different activities in the input-output model: households, restaurants and in canteens in various activities. All inputs of food to households and restaurants are regarded as being purchased for being consumed. For canteens which are present in basically all activities or industries in the input-output table, care should be taken; some of the inputs of food is used as raw material for the manufactured products in these activities. Some example are: when the chemical industry uses vegetable oil as precursors for surfactants; when oil mills use oil crops for the production of vegetable oils; when animal farms use grains and other food this is used as animal feed; and when oil refineries uses vegetable oil and crops for biodiesel and bio-ethanol. Therefore, a number of inputs of food in some activities are not accounted as being for food consumption. This is especially relevant for agriculture, the food industry, the chemical industry and the oil refinery industry. When all food inputs for consumption are identified, the next step is to estimate how much of this food is actually consumed, i.e. digested by humans. For this purpose, the food consumption data from FAOSTAT (2013) and human metabolic balances (Muñoz et al. 2007). The difference between the use of food in the food consuming activities (households, restaurants and canteens) and the digested food is regarded as food waste. All generated food waste is sent to treatment: landfill (83%), incineration (11%), composting and biogasification (6%); the treatment mix is based on an average of national mixes, the data sources are further described in Merciai et al. (2013). It should be noted that the emissions from landfills in some developing countries are associated with large uncertainties; known figures on landfilled are modelled as managed landfills whereas the rest has been modelled as non-registered waste where there are no other emissions than the CO_2 from decay of organic material. The faeces/urine for wastewater treatment is determined by the human metabolism balance. Also here the emissions in some developing countries are associated with large uncertainties.

Since the functional unit concerns the consumption of food, also the energy, water, cleaning chemicals, domestic machinery, and transport for shopping in the households that relate to food consumption need to be included. The Exiobase v2 database contains information on the total use of these inputs in households. The shares of these inputs that relate to food consumption are estimated using household accounts in Schmidt (2010). For each input, it has been assured that the same quantity is also sent to waste treatment (landfill, incineration, recycling etc.) based on global average of national waste treatment mixes, e.g. when 1 kg domestic machinery is used for food preparation in households, the waste treatment of 1 kg domestic machinery is also included.

3. Results

The results chapter is introduced by a presentation of the outcome of the mass flow analysis of the world's food consumption. After this, the GHG-emissions and land-use related to the food consumption are presented.

3.1. Mass flow analysis results of the world's food consumption

The starting point of the life cycle assessment was a detailed mass flow analysis of all food in the world. The functional unit: 'the world's food consumption in 2007' is specified in details in Table 1. The total food consumption was divided into 25 food items and three different activities where food is consumed: canteens (including events), restaurants, and households. It should be noted that the total food consumption at 2366 million tonne (dry matter) is the input of food to the food consuming activities, i.e. the input of crops, animal products and food items upstream are larger because losses occur along the product chain. The use of food was divided into food from three different sources: crop production, animal production, fishery, and the food industry. 42% of the food was sourced directly from crop production, 3% from animal production, 1% from fishery, and 55% from the food industry. 10% of the food was meat (beef, pig, poultry, fish and other), 5% was other animal products (milk, egg etc.), and 85% was non-meat. It can also be derived from Table 1 that 6% of the world's food was eaten in canteens (industry, schools, at sport events etc.), 14% was eaten in restaurants, and 80% in households.

Table 1. Functional unit: Global food consumption in 2007. Flows are given in million tonne dry matter (except last two columns).

Food item	**Food use in canteens**	**Food use in restaurants**	**Food use in households**	**Total**	**Kg dry matter per capita**	**Moisture content**
Crops: Paddy rice	5	6	74	**85**	13	15%
Crops: Wheat	13	8	82	**104**	16	14%
Crops: Cereal grains, other	16	9	54	**80**	12	15%
Crops: Vegetables, fruit	6	9	459	**474**	71	82%
Crops: Oil crops	9	10	40	**60**	9	9%
Crops: Sugar cane/beet	4	109	3	**115**	17	75%
Crops: Crops, other	2	3	64	**69**	10	79%
Animals: Cattle, beef	0	0	0	**0.5**	0	53%
Animals: Cattle, raw milk	2	2	2	**7**	1	88%
Animals: Pigs	1	1	9	**11**	2	55%
Animals: Poultry	15	0	7	**22**	3	70%
Animals: Meat animals, other	0	1	2	**3**	0	57%
Animals: Animal products, other (e.g. egg)	2	2	16	**20**	3	26%
Animals: Fish	0	1	10	**12**	2	80%
Food: Meat, cattle	2	5	24	**31**	5	61%
Food: Meat, pigs	3	15	53	**71**	11	41%
Food: Meat, Poultry	1	6	31	**38**	6	75%
Food: Meat, other	0	1	8	**10**	1	35%
Food: Vegetable oils	11	25	36	**71**	11	0%
Food: Dairy products	7	10	70	**87**	13	46%
Food: Rice, processed	3	10	110	**123**	18	15%
Food: Sugar	7	50	177	**234**	35	1%
Food: Food, other	17	44	475	**536**	80	33%
Food: Beverages	13	10	45	**68**	10	90%
Food: Fish products	3	3	29	**35**	5	80%
Total	**143**	**341**	**1,881**	**2,366**	**355**	**43%**

Based on the balanced mass flows of resource inputs, products and emission outputs in the CREEA model, a quantitative illustration of the product system relating to the world's food consumption is created. Only the activities involved in agriculture, fishery, food industries and food consumption are shown. The actual modelled product system includes 166 activities in total. The product system is presented in Figure 2.

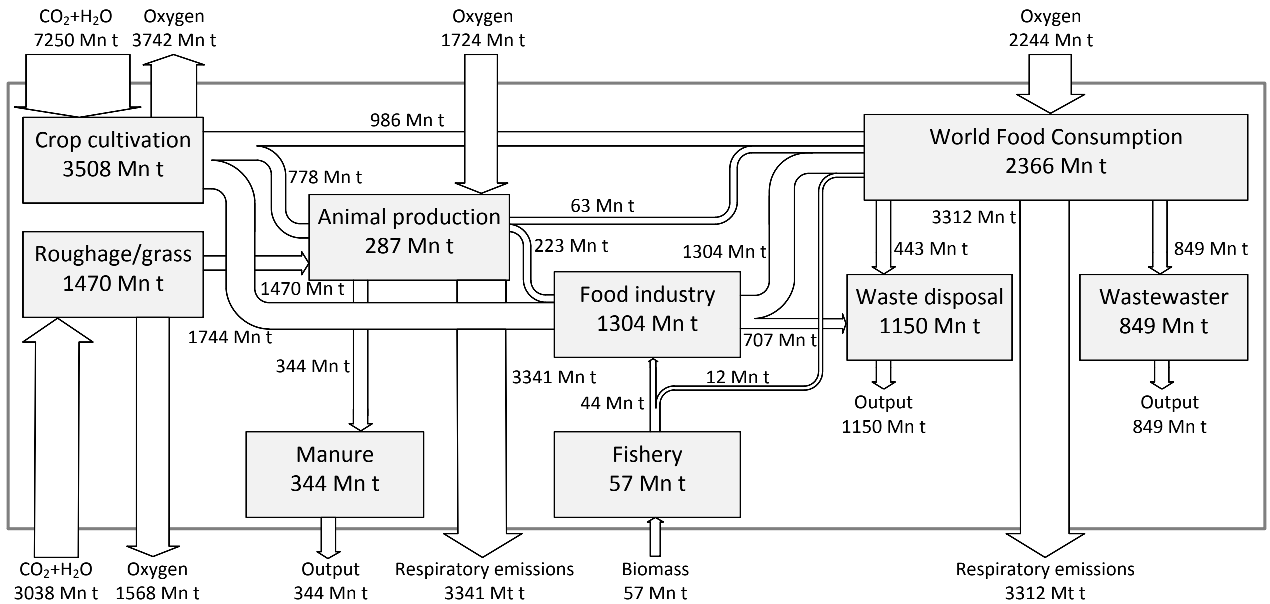

Figure 2. Mass balanced product system relating to the functional unit: the world's food consumption. The boxes represent activities, the arrows represent flows, and the grey borderline illustrates the system boundary. The numbers in the boxes show the total product output from the activities. All flows are given in million tonne (Mn t) dry matter.

It appears from Figure 2, that the largest flows on the input side were crop cultivation and roughage/grass. Of the total plant material produced, the largest share was used as animal feed; 45% of all plant material was used as input for animal production, and the remaining plant material was crops to food industry (35%), and to food consuming activities (20%). The largest share of cultivated crops were inputs to the food industry; 50%. The remaining crops were used as animal feed (22%), and directly for food consumption (28%). Waste or materials for treatment flows occurred in animal production (manure), food industry (food waste), and in the food consuming activities (food waste to disposal and faeces/urine to wastewater). The largest material for treatment flow was the faeces/urine to wastewater (849 million tonne). Another major material for treatment flow occurred in the food industry (707 million tonne), but also large quantities of food wastes occurred in the consumption activity (443 million tonne). It should be noticed that most of the materials for treatment in the industry are utilized for various purposes. However these materials for treatment or by-products are currently not modelled in detail in the Exiobase v2 model.

The most remarkable highlights from Figure 2 are:

- Of the total plant material produced/extracted for food purposes at 4978 million tonne, only 48% (2366 million tonne) ended up as an input to the food consuming activities. And further in the food consuming activities, additional 443 million tonne was food waste. Hence, only 39% of the raw material for food ended up being consumed.
- Almost half (45%) of all the plant material produced/extracted was used as animal food, while animal products only account for 15% of the input of food to the food consuming activities (see Table 1).

3.2. Life cycle impact assessment (LCIA)

This section presents the life cycle impact assessment of the world's food consumption. The results are shown as GWP100 for GHG-emissions (IPCC 2007) and in addition the land-use (occupation) is also described. Global GHG-emissions, measured as GWP100, in 2004 were around 50,000 million tonne CO_2-eq. (IPCC 2007).

Equivalent data for 2007, which was the basis year for the LCA, have not been identified. This will likely be published as part of IPPC's fifth assessment report in 2014. Therefore, when comparing the food consumption related emissions with global emissions, reference is not made to the same year, and hence these comparisons should be interpreted with care since the GHG-emissions in 2007 would be higher than 2004.

The LCA showed that the total impact on GHG-emissions from food consumption in 2007 was **25,370 million tonne CO_2-eq.** Compared to global GHG-emissions in 2004, food consumption in 2007 accounted for approximately 50% of all GHG-emissions. When also including the contribution from indirect land-use changes (iLUC), the emissions increased by 16% to **29,450 million tonne CO2-eq.** This corresponds to almost 60% of global GHG-emissions in 2004. The total land-use was **4,900 million ha** which corresponds to 38% of the global land area. This was distributed on 25% cropland, 7% roughage/intensive grass and 69% extensive pastures (grassland).

Two different contribution analyses are presented: In Figure 3, the overall result (without iLUC) at 25,370 million tonne CO2-eq. is broken down in a contribution analysis which shows the contributions in terms of the impact of inputs to the food consumption activities (canteens, restaurants, and households). Figure 4 breaks down the overall result in terms of the direct emissions by each activity in the product system.

It appears from Figure 3 that 36% of the total impact was related to the inputs of meat. When including also other animal products (dairy, egg), then this increased to 51%. The use of processed food accounted for 22% and non-processed food (raw food) for 8%. The contribution from energy (8%) relates to the electricity and fuels used for storing, preparing and dish washing in the food consuming activities. Waste disposal (5%) relates to the impacts from the food and waste water generated in the food consuming activities only. The impact from upstream food waste was included in the results for the input of food products to the consuming activities.

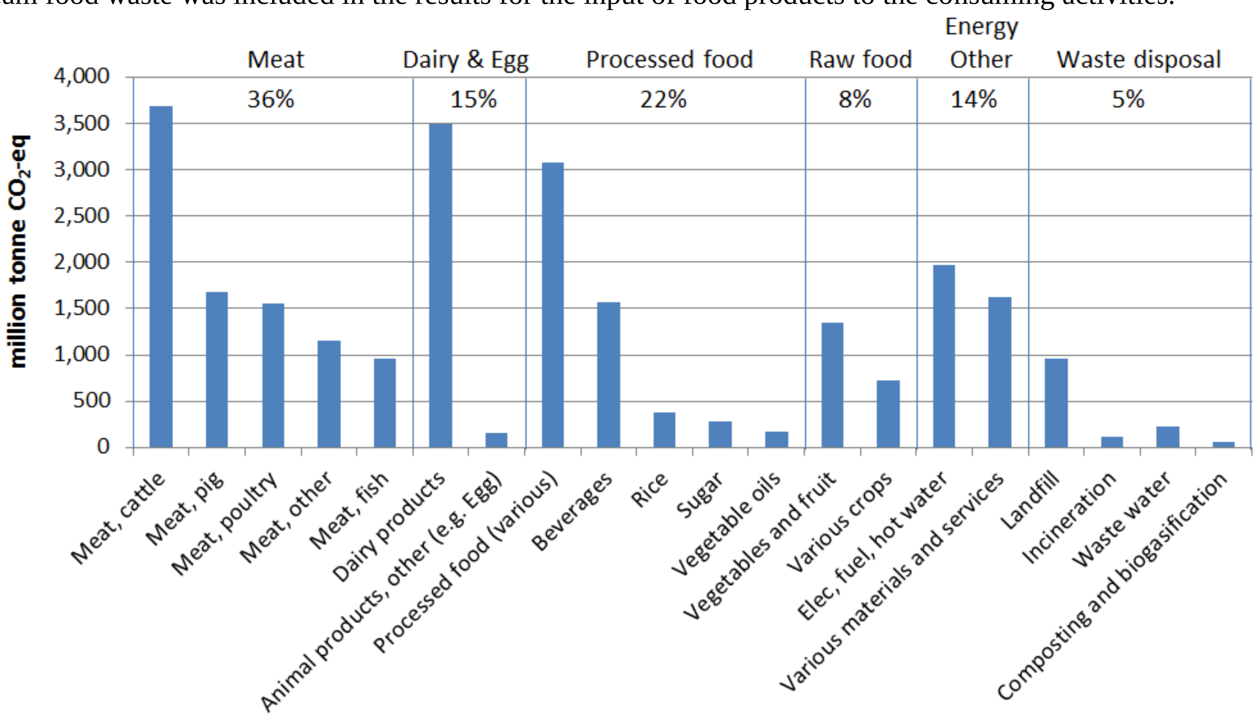

Figure 3. Contribution analysis based on the direct inputs to the food consuming activities. The results do not include the contribution from iLUC.

Figure 4 shows that 32% of the total emissions at 25,370 million tonne CO2-eq. took place in the animal producing activities in agriculture. Note that the cultivation and maintenance of roughage and grass areas were included in the animal activities. 7% of the GHG-emissions took place in crop production. It also appears from Figure 4 that energy was a hotspot in the food system. Around 14% of all food related emissions took place in activities where fuels are produced, converted to energy or transmitted to the energy user. Fertiliser production accounted for 6% and 2% were emitted in the food processing industries. The latter included combustion of fuels and process emissions (e.g. CH_4 from palm oil mill effluent treatment). 7% of the emissions took place in waste treatment activities. The remaining 32% 'other' relates to the production of packaging, machinery, transport, wholesale and retail etc.

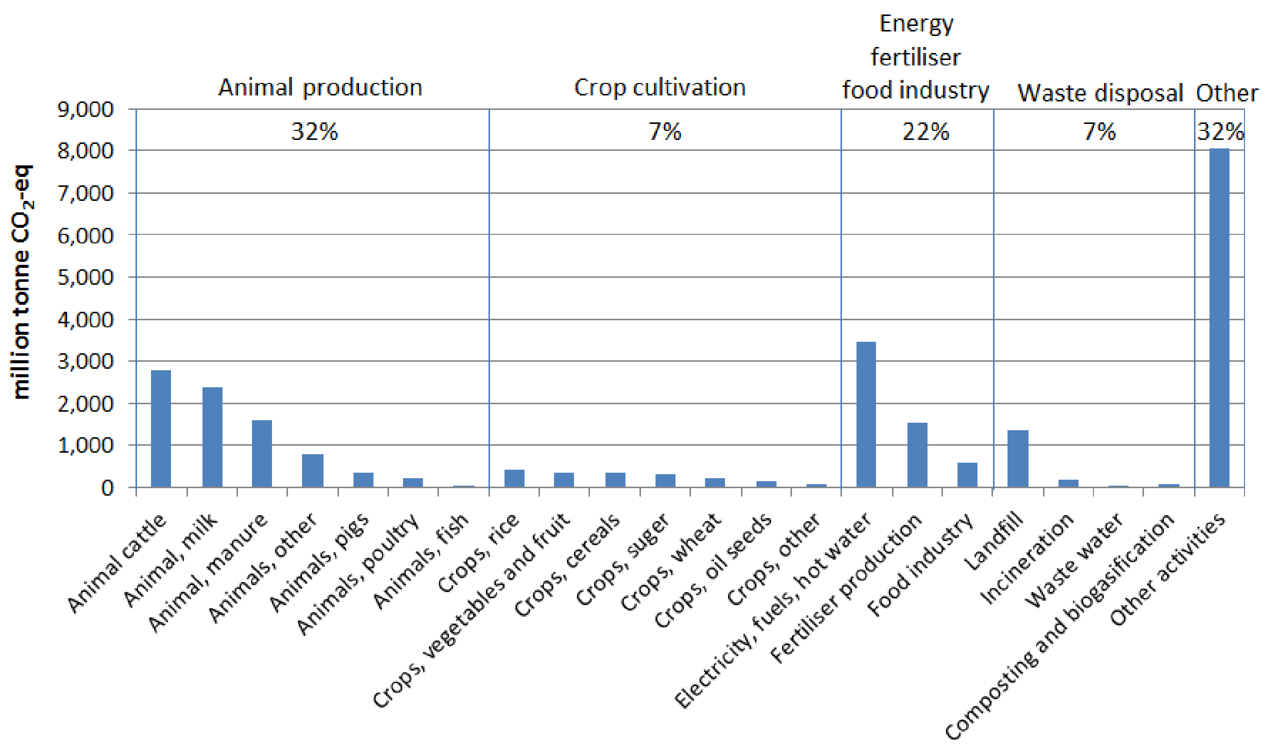

Figure 4. Contribution analysis based on direct emissions of individual processes in the product system. The results do not include the contribution from iLUC.

Figure 5 shows the GHG-emissions per kg dry matter of food product – without and with iLUC. The figure clearly shows that the impacts from animal products were significant higher than non-meat products. The reason is that animals need 4-40 kg dry matter feed per kg dry matter animal product depending on the type of animal. The most feed-to-product efficient animals are poultry, pigs, and fish, while meat production in the cattle and sheep systems (ruminants) is much less efficient. Milk production is somewhere in between. In addition to the relative low feed-to-product efficiencies of animal system compared to non-animal systems, the animals also produce emissions themselves; especially ruminants where methane from enteric fermentation is high. It can also be seen in Figure 5 that processed food was associated with higher impacts than non-processed food. The difference was related to losses of food waste in the processing as well as energy inputs. When including iLUC, the GHG-emissions increase by approximately 40-50% for crops, 10-30% for animals (smaller for fish), and 10-30% for processed food.

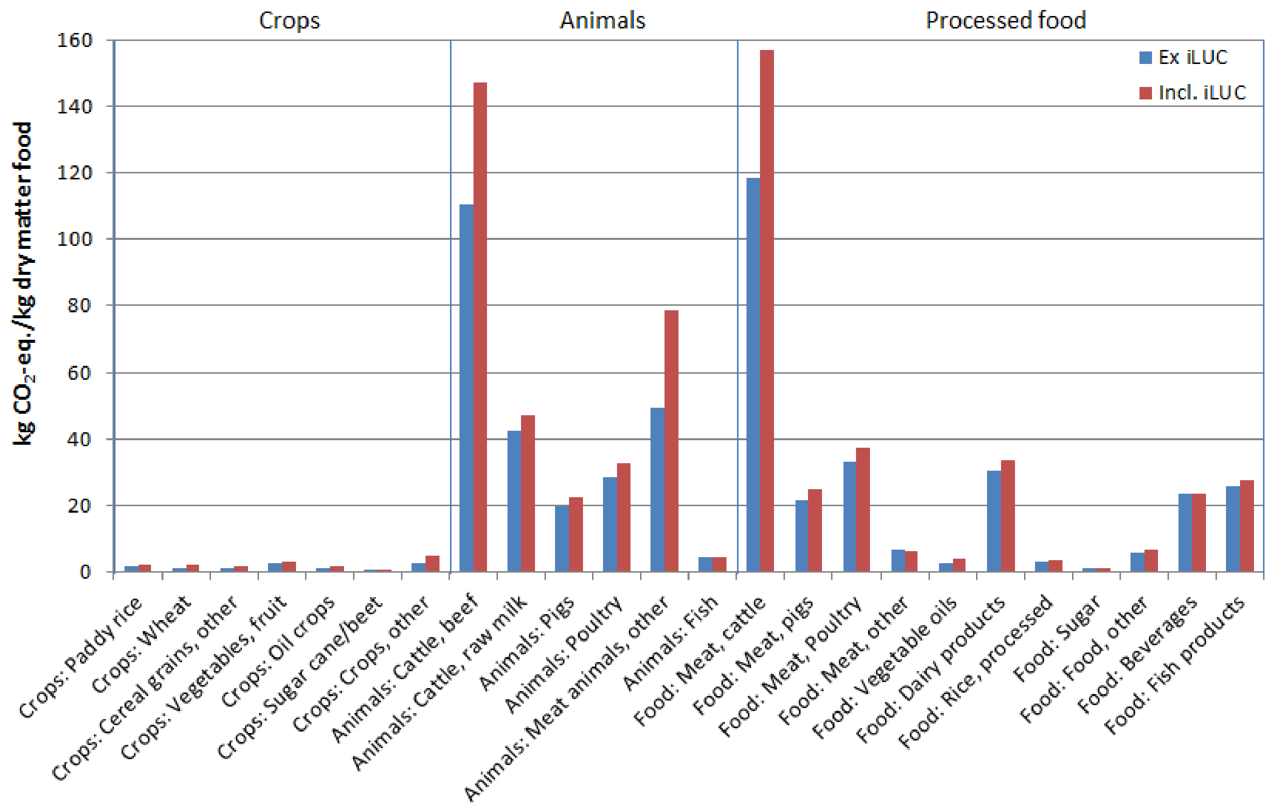

Figure 5. GHG-emissions per kg dry matter product. The results are shown with and without the contribution from indirect land-use changes (iLUC).

4. Discussion

This paper presented the use of a new model to create a detailed mass flow analysis and LCA of the world's food consumption. Since the model was based on several inherent mass balances, the level of consistency and accuracy of the results are regarded as being high. However, the presented results were based on a preliminary version of the Exiobase v2 model. Therefore, the final revisions of the model may also lead to changes in the results presented in this paper.

Since the model has a global scope and a national level of detail, it is very useful for analyzing various consumption groups at different levels (national, regional, global) as well as specific products or industries. The integration of mass flow analysis and input-output life cycle modeling in a hybrid unit database also show advantages; the mass unit of products in the database makes it look very much like a traditional process database in LCA software. Further, the mass flow data are useful for creating mass balanced product systems, and to obtain national and global scale overview of industries or consumption groups. Also, the combination with monetary input-output models ensures that emissions and product flows from all activities in economy are included.

The presented results were on a global scale, and at the level of detail which was allowed from the product classification in the Exiobase v2 model. The model immediately allows for similar analysis at the national level, and with some limited additional effort, it is possible to achieve a higher level of detail in terms of included food product categories to any desirable level. Further, similar analyses can also be carried out for any other consumption group than food, e.g. paper, electronics, waste etc. Hence, this article only represents an initial glimpse of the opportunities for analyses with the Exiobase v2 model.

5. Conclusion

The purpose of this paper was to carry out a mass flow analysis and an LCA of the world's food consumption. For this purpose a preliminary version of the novel model Exiobase v2 was used. As an add-on to the Exiobase model, a model of indirect land use changes (iLUC) was integrated with the Exiobase v2 model. The results were shown without and with the contribution from iLUC. The total food consumption in 2007 was 2366 million tonne (dry matter). 10% of the food was meat (beef, pig, poultry, fish and other), 5% was other animal products (milk, egg etc.), and 85% was non-meat. 6% of the world's food was eaten in canteens (industry, schools, at sport events etc.), 14% in restaurants, and 80% in households. The total plant material produced/extracted for food purposes was 4978 million tonne. Of this only 48% (2366 million tonne) ended up as an input to the food consuming activities. And further in the food consuming activities, additional 443 million tonne was food waste. Hence, only 39% of the raw material for food ended up being consumed. Almost half (45%) of all the plant material produced/extracted was used as animal food, while animal products only accounted for 15% of the input of food to the food consuming activities.

The LCA showed that the total impact on GHG-emissions from food consumption in 2007 was 25,370 million tonne CO_2-eq. Compared to global GHG-emissions in 2004, food consumption in 2007 accounted for approximately 50% of all GHG-emissions. When also including the contribution from indirect land-use changes (iLUC), the emissions increased by 16% to 29,450 million tonne CO2-eq. The total land-use was 4,900 million ha*year which corresponds to 38% of the global land area. This was distributed on 25% cropland, 7% roughage/intensive rotation grass and 69% extensive pastures (grassland). Of the total impact at impact at 25,370 million tonne CO_2-eq. (excluding iLUC), 36% was related to the inputs of meat. When including also other animal products (dairy, egg), then this increased to 51%. The use of processed food accounted for 22% of the GHG-emissions and non-processed food (raw food) 8%. The contribution from energy related to the electricity and fuels used for storing, preparing and dish washing in the food consuming activities accounted for 8%. Waste treatment of the food waste generated in the food consuming activities accounted for 5%. The remaining 6% related to domestic machinery, cleaning chemicals etc. As for the mass flow analysis, the LCA also revealed remarkable impacts related to meat and animal product consumption; while animal products only accounted for 9% of the food input to food consuming activities, the GHG-emissions from these animal products accounted for 51% of the food related GHG-emissions.

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Life Cycle Sustainability Assessment of Dairy Farming at the Grignon Farm

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ABSTRACT

A Life Cycle Sustainability Assessment of dairy production system at the Grignon farm is presented. Two systems are studied, corresponding to the feed mix used before 2008, and the feed mix used thereafter, including a higher share of corn silage, alfalfa, locally produced rape meal and grazing; the functional unit is 1 kg standard FPCM at the farm gate. An innovative approach to assess impacts on biodiversity and soil quality is included, next to relevant environmental LCA impact categories, including water use and land use, as well as social and economic sustainability indicators. Comparison between previous and current diet shows a reduction in global warming potential and cumulative energy demand, two main objectives of the farm. Benefits and trade-offs with regard to other impact categories are presented and discussed, and recommendations are derived for further optimization.

Keywords: Life Cycle Sustainability Assessment, Dairy, Biodiversity,

1. Introduction

1.1. Overview and Context

The experimental farm of Grignon is a mixed crop-livestock farm. It is located at a distance of 40 km from Paris. The farm hosts 120 Prim'Holstein dairy cows with a 30 liter average daily milk production, in two milking times, which represents ca. 1,100,100 liter production every year on the farm. The agricultural net area of the Grignon farm exceeds 500 ha. This surface is dedicated to pasturelands and fodder and cereal production. The main crops are wheat, fodder barley, beer barley, canola, corn, horse bean, alfalfa, and meadows.

In order to reduce the farm's energy consumption and GHG emissions, there has been a change in the feed for dairy cows around the year 2008, which affected the composition of the feed mix. In the following, the new feed composition used since then will be referred to as 'current diet', whereas the alternative feed composition will be referred to as 'previous diet '.

In line with the objectives of the farm there is a need to investigate the consequences of this change in diet on greenhouse gas emissions and energy consumption but also in other potential impacts on sustainability in a wider context, encompassing environmental, social and economic aspects. Therefore, the farm and BASF have conducted together a Life Cycle Sustainability Assessment (LCSA) based on the AgBalance methodology created by BASF (Schoeneboom et al. 2012)

The present study hence quantifies and compares the sustainability performance (based on Life Cycle Assessment and additional sustainability criteria) of dairy (milk) production at the Grignon farm based on the current and the previous dairy cow feed mix.

A critical review has been carried out by a panel of interested parties, represented by external experts.

1.2. Objectives of the Study

The main objective of the study is to investigate sustainability performance of dairy production system at the Grignon farm by quantifying key environmental, social and economic performance characteristics (indicators). Due to the farm's objectives, the focus is on climate change impact and energy consumption.

Sub goal is to investigate the sustainability impact of the feed composition used at Grignon farm, specifically the effect of a change from the feed composition around the year 2008 (previous diet) to the feed used since then (current diet).

The intended audiences of the study include stakeholders of the farm and other interested parties, including the general public.

2. Methods

2.1. Studied Systems and Functional Unit Definition

The systems chosen for this comparative analysis are the dairy cow production system at the Grignon farm with the typical feed composition in the period since 2009, referred to as ‘current diet’, and the dairy cow production system based on the feed composition before 2008, referred to as ‘previous diet’, respectively.

Table 1. Diet composition of current and previous diet for one dairy cow life in absolute quantities (dry weight)

Feed component	System 1: Current diet Quantities [kg dw / cow life]	System 2: Previous diet Quantities [kg dw / cow life]	Origin
Maize silage	10,214	3,815	Grignon farm
Beet pulp	2,818	4,419	market
Alfalfa silage	1,916	0	Grignon farm
Rape meal from Grignon	4,428	0	Grignon farm
Alfalfa hay	3,252	0	Grignon farm
Grain maize	1,929	4,134	Grignon farm
Hay	2,499	2,593	Grignon farm
Molasses	1,851	2,451	market
Grazed grass	1,855	347	Grignon farm
Rape meal, industrial	1,421	3,186	market
Orange skin	243	307	market
Minerals (CaCO3 + NaCl)	618	1,192	market
Wheat straw	437	1,610	Grignon farm
Dried distillers grain	352	560	market
Humid distillers grain	0	1,470	market
Dehydrated alfalfa	4	1,662	market
Horsebean	0	881	market
other	1,140	2,252	market

Table 1 lists the composition of the feed mix for both systems, giving totals of each feed component aggregated over all growing stages of the cows’ life (veils, heifers, dairy cows). The comparison shows that the dominating component in the current diet is by far maize silage (44%) produced on the farm. On the other hand side, previous diet is composed by 24% of sugar beet pulp (purchased from a supplier) and 20% maize silage. ‘Rape meal industrial’ use was reduced significantly in the current diet and is now just 45% of the previous diet (compare Table 1). Rape meal industrial refers to the residual by-product of rapeseed oil pressing, which is purchased by Grignon from the general market. In the current diet, a new type of rape meal was introduced. This rape meal is obtained from a local oil mill, that uses oilseeds from rape produced at the Grignon farm, and has higher residual oil content than the industrial rape meal. Herein, we will refer to this new type of rape meal as "Rape meal from Grignon". The rape meal from Grignon replaces industrial rape meal in the current diet to a large extent; the remaining fraction of industrial rape meal comes from the ready-mixed feed products which are still used in the current diet. Another difference between the systems is the amount of grazed grass which increased by 300% in the current diet.

Dairy cow systems provide several functions: milk, veil (for fattening), manure, and the products derived from slaughter of the cows. The relevant function of the system considered herein is the provision of milk. Allocation is used to resolve multi-functionality (see below).

The functional unit provides a basis for comparing all life cycle components on a common basis and allows direct comparisons among the product systems in question. The systems are compared based on the functional unit of 1000 kg standard fat and protein corrected milk (FPCM).

A simplified representation of the processes considered in the present study is shown in Figure 1. Although not shown, the calculation considers all activities ‘upstream’ from the extraction of raw materials to manufacturing of basic intermediate products etc. and including transportation. The study considers process up to the farm gate; further activities 'downstream' as distribution, consumption and end-of-life of the milk are not taken into account. This is usually referred to as "cradle-to-gate" study.

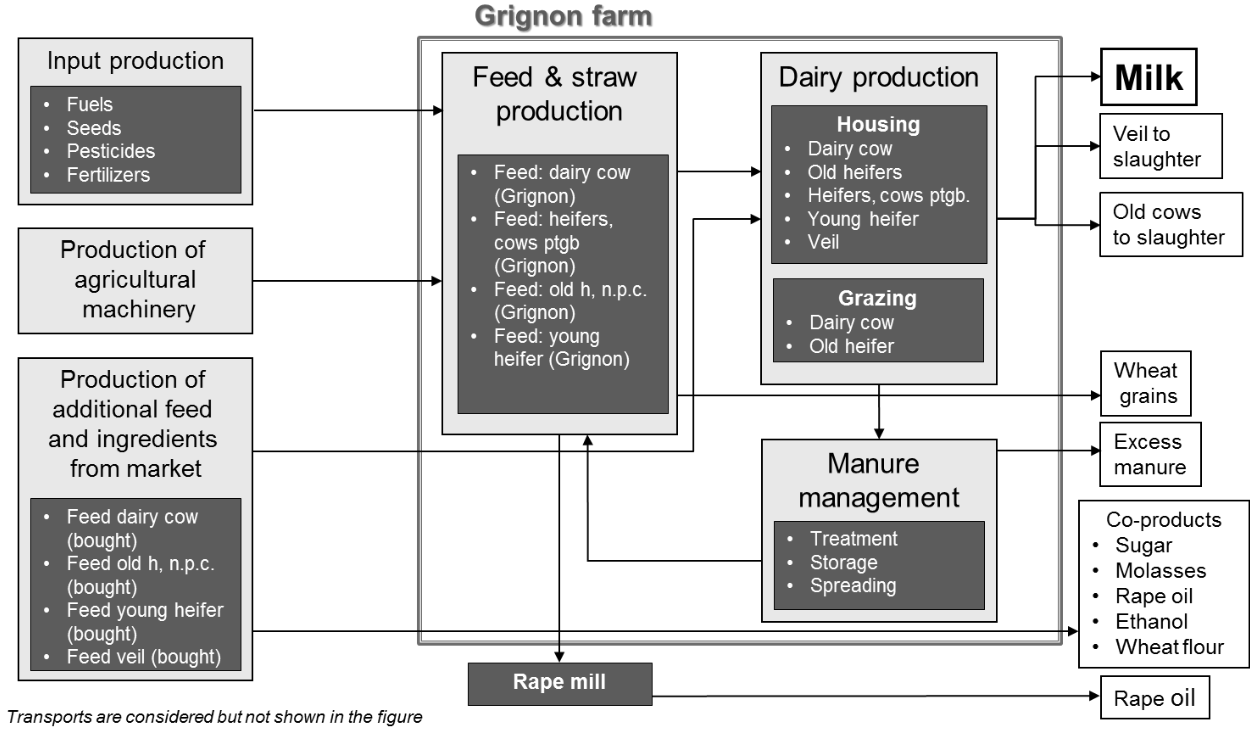


Figure 1. Generic system boundary diagram for current and previous diet.

2.2. Inventory Methods and Assumptions

To achieve the objective of the study attributional life cycle inventory method is used. A consequential LCA was considered to be not necessary for the objectives and intended use of the study, because it is aimed at a retrospective or descriptive comparison of milk production, the system is thus modeled as it was; average or generic data is used. Conclusions of this study do not serve as a basis for long-term policy decision making.

Feed components:

All relevant feed components have been considered, either by collecting data from the Grignon farm, or by consulting relevant databases for appropriate life cycle inventory datasets (see below). Feed components where data was not readily available were neglected when their estimated impact falls below 1% of the total system.

For the feed components which are purchased from an external supplier, life cycle inventory data from the ecoinvent database (version 2.2) was used (ECOINVENT 2007). In a few cases, for example in the case of dehydrated alfalfa and orange skin, an equivalent inventory was not available, and an approximation was taken instead. The datasets for inputs needed to model the production activities at the Grignon farm have all been taken from the ecoinvent database (version 2.2).

Primary data for feed components produced on the Grignon farm, as well as for the dairy cow farming and manure management was collected in a dedicated data collection sheet. The social data for the other processes, i.e., all activities not carried out on the farm, are derived from statistical data for industry sectors (Kölsch et al. 2008). The economic costs are calculated at the production step of the milk, i.e., from the perspective of the farm. Therefore, cost data has only been raised from the Grignon farm; no costs therefore had to be obtained for upstream processes.

The multi-output system 'dairy cows' produces milk, excess veil, dairy cows to slaughtering, and excess manure. Both economic allocation as well as the procedure proposed by International Dairy Federation (IDF 2010) for allocation were tested and lead to identical allocation factor for milk of 87%. To ensure consistency between ecoinvent and Grignon specific feed component datasets the same allocation factors are used. For the following multi-output processes economic allocation was applied (allocation factor to co-products derived from ecoinvent documentation in brackets): Feed components: co-products include rape meal (25.7%), beet pulp (3.8%), distillers grain (2.3%) and straw (7.5%).

Excess manure produced by the livestock and not used within the system (for arable production of fodder crops) was treated as substituting an equivalent amount of N, P_2O_5 and K_2O from a mineral fertilizer production process.

Emissions:

CH_4 emissions from enteric fermentation as well as from manure management are calculated based on IPCC 2006, according to the 'tier 3' approach. N_2O, NH_3 and NO emissions from manure management are calculated based on IPCC 2006, according to the 'tier 2' approach.

Emission factors for direct and indirect field emissions of N_2O and CO_2 are taken from IPCC 2006 (tier 1). CO_2 emissions due to land use change (according to IPCC 2006) are not considered because no production area was converted to arable land in the past 20 years.

Water:

The inventory includes consumption of water over the complete product system. Water used in the cow housing system is considered to be 100% consumptive because this water is either diverted to milk, to slurry and manure or consumed otherwise by the animals. The fraction of water applied to agricultural fields with slurry is considered to be transferred to other watersheds and lost through evapotranspiration. In crop production, all forage crops are non-irrigated. Water consumption in industrial manufacturing of fertilizers was modeled based on the following assumptions: 1% of cooling water is considered consumptive (99% returned to the river), based on actual data of the BASF Ludwigshafen production site. As for the water consumption of other water resources we followed the recent study of De Boer et al 2013; these authors included consumptive water use based on ecoinvent: sea water and turbine water (in-stream use) were excluded from the inventory. All other water sources (lake, river, well, unspecified) were assumed to be consumptive. The results were taken from this study and amount to the following numbers: 4 L per L Diesel, 1.3 L per kWh electricity, 4.6 L per ton of concentrate produced at the feed mill, 0.652 L per ton-km road transport, 0.16 L per ton-km for inland ship transport, 0.029 L per ton-km transoceanic ship transport.

In line with the goals of the present study, which are to compare the sustainability of the feed compositions and the contribution of the feed composition to the overall impact, several parameters were assumed to be identical in both product systems: (i) the same milk productivity is assumed for both product systems (10.475 litres per lactation period); (ii) technical itineraries and yields for crop production are assumed to be identical in both systems; (iii) prices for inputs (e.g. fertilizers, pesticides) were held constant for both systems, to average 2012 prices.

2.3. Life Cycle Impact Assessment Method

The environmental assessment is based on established life-cycle impact assessment (LCIA) mid-point indicators as used also in Eco-Efficiency Analysis (Saling et al. 2002) and a wide range of other LCA approaches. Environmental impact categories are: abiotic resource depletion (Saling et al. 2002), climate change (IPCC 2007), acidification (CML), photochemical ozone creation (CML), fresh water eutrophication (Goedkoop et al. 2013), marine eutrophication (Goedkoop et al. 2013), water use related to water scarcity (Pfister et al. 2009), ecotoxicity (Rosenbaum et al. 2008), land use (EDP; Köllner and Scholz 2007, Köllner and Scholz 2008). Primary energy consumption (Saling et al. 2002) was assessed to meet the farm's objective to measure and reduce energy demand.

Additionally, environmental indicators addressing the specific impacts of agricultural activity on biodiversity in agricultural areas, and on soil health and conservation, have been incorporated into the methodology (see Schoeneboom et al, 2012 and references therein). The potential impact on biodiversity is assessed using eight indicators relating to the state of biodiversity, crop management practice (e.g. crop rotation, ecotoxicity potential of crop protection products, nitrogen surplus) and the environmental policy context (e.g. availability of protected areas or agri-environmental schemes). The impact category of soil health is comprised of indicators assessing soil organic matter balance, soil erosion, soil compaction and the nutrient balances of N, P, K.

In terms of economic assessment, both production costs as well as economic performance are taken into account. Production costs are grouped into variable and fixed costs and quantified using an overall total cost of ownership for the defined functional unit (Kicherer et al. 2007). Economic performance is assessed using indicators for farm profitability as the central criterion for economic sustainability, subsidies which may excert distorting economic effects and productivity as measured as the production value of agricultural goods per hectare weighted by the contribution of the agricultural sector to the national GDP.

The social assessment in AgBalance is derived from the SEEBALANCE method for social LCA, which was developed in 2005 by Universities of Karlsruhe and Jena, the Öko-Institut (Institute for Applied Ecology) Freiburg e.V., and BASF (Schmidt et al. 2005, Kölsch et al. 2008). Based on the UNEP-SETAC guidelines for social LCA of products 5 stakeholder categories were defined: Farmer, consumer, local community, internal community and future generations. The SEEBALANCE indicators and data sources are employed to assess the social impacts of industrial up- and downstream processes. For the agricultural activities in the life cycle, a set of adapted social impact indicators was integrated into the AgBalance method which was designed to match closely the same social sustainability topics addressed in the assessment of the upstream and downstream processes.

3. Results

3.1. Environmental Impact Assessment

The contribution analysis of the environmental impacts below makes use of the following category definitions:

(i) emissions from manure management

(ii) emissions from livestock enteric fermentation

(iii) ancillary input for cow housing: contains among others diesel (production and combustion) needed for supplying and distributing the feed to the cattle on the farm (e.g. extraction of silages from the silo and on farm transports). Furthermore electricity for lighting, pumping of slurry etc. is included. Other materials combined in this category are: consumed blue water, wheat straw for bedding and farm infrastructure (milking parlour and housing system)

(iv) feed production: aggregated impacts due to production of agricultural inputs and field emissions from fodder crop production

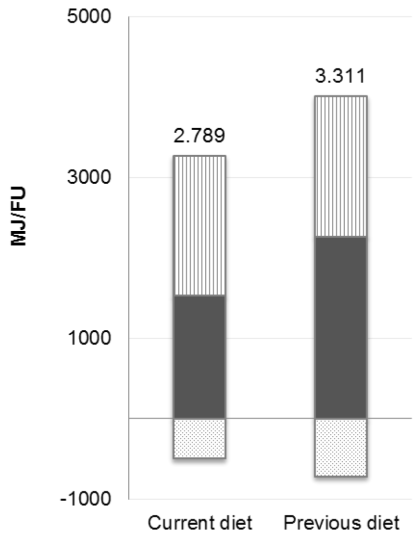
(v) avoided fertilizer production: Excess manure produced by dairy cows and not used within the system (for arable production of fodder crops) was treated as substituting an equivalent amount of N, P_2O_5 and K_2O from mineral fertilizer production processes. This contribution appears as a negative impact in the graphs.

Because the objectives of the farm are to reduce energy consumption and greenhouse gas emissions, we focus on these two impact indicators first.

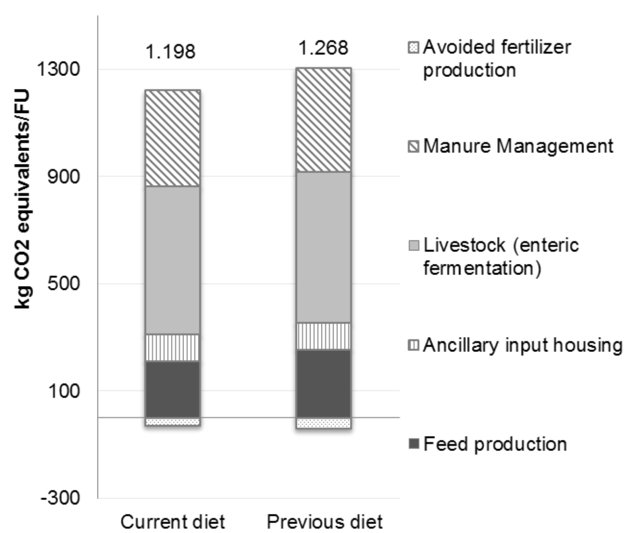
Figure 2 shows the primary energy consumption (excluding biomass feedstock energy for feed) for the current and previous diet calculated excluding biomass feedstock energy in feed. Primary energy consumption of the milk production for 1000 kg FPCM with the current feed mix is 2.8 GJ and 16% reduced in relation to the previous diet. The dominating contributions arise from the diesel and electricity use in housing and the feed production. This in turn is dominated by fuel and mineral fertilizer production and use.

It is noted that the current diet is associated with a higher input of biomass total matter (cf. Table 1), mainly of maize silage from the Grignon farm. If the biomass feedstock energy of the fodder is taken into account (which stems from sun energy), the current diet system exhibits a significantly higher energy consumption of 14 GJ/FU.

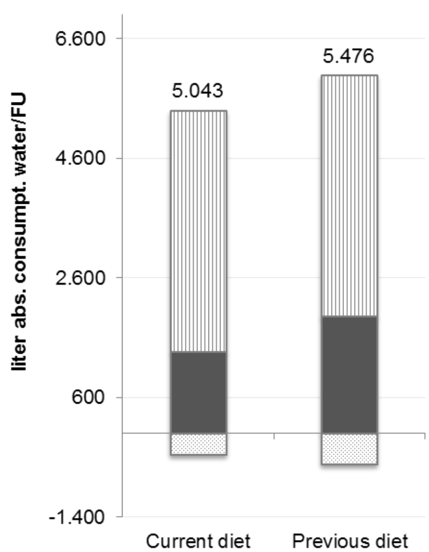
The global warming potential for both systems is shown on the right side of Figure 2. The current diet exhibits a 5.8% lower global warming potential (GWP) compared to the previous diet. The largest contribution to this improvement in GWP of the new diet (i.e., the difference between both systems) comes from the feed production: there is a significantly lower GWP associated with the feed in system ‘current diet’ than in system ‘previous diet’ (current: 214 kg CO_2e/FU, previous: 257 kg CO_2e/FU, 17% reduction). In more detail, this difference from feed production can be attributed particularly to the dairy cow feed. In sum, the GWP attributed to dairy cow feed is reduced from 223 kg CO_2e/FU in system ‘previous diet’ to 175 kg CO_2e/FU in system ‘current diet’. This net reduction in GWP is because of the comparably low GWP impact of the production of the new feed components, particularly those produced on the Grignon farm. An inspection of the corresponding input data of these inventories (not shown here) reveals that in oilseed rape, maize, and hay production at Grignon, mixed organic and mineral fertilization is applied, thus reducing mineral fertilizer use. Furthermore, alfalfa, a leguminous (N-fixing) crop, is grown without any addition of mineral N-fertilizer.



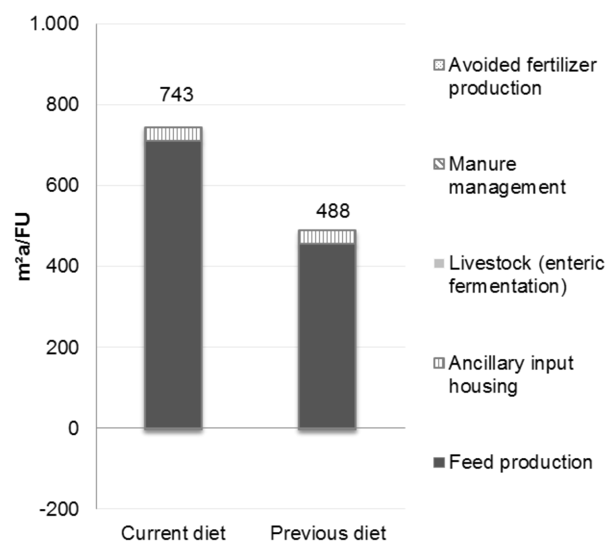
Primary energy consumption excluding biomass feedstock energy of feed



Climate Change impact



Water use impact related to water scarcity



Land use impact (EDP)

Figure 2. Primary energy consumption excluding biomass feedstock energy of feed (left) and climate change impact (right) of the current and previous diet.

The second largest contribution to the reduction in GWP with the current feed is due to changes in methane emissions from manure management. With the 'current diet', older heifers and dairy cows are fed in a mixed housing (liquid/slurry and solid manure storage) and grazing (pasture) system, whereas the 'previous diet' system used exclusively housing (liquid/slurry and solid manure storage). The decisive factor in the present IPCC modeling approach is the 'methane conversion factor' for each manure management system (MCF). This factor is more favorable for pasture, than for solid manure storage, and much more favorable than liquid/slurry storage. In simpler words, the higher the share of pasture in the feed system is, the lower the methane emissions from manure management are.

Analyzing the contribution of each aspect to the total GWP of each system, it is apparent from Figure 2 that livestock emissions of methane from enteric fermentation represent the highest share, followed by methane emissions from manure management. Feed production then itself makes the third largest contribution to the total GWP

Water Use: The impact of water consumption related to freshwater deprivation (water scarcity) is assessed over the cradle-to-gate life cycle of the milk. Forage crop production in the present study does not make use of irrigation; therefore the main contribution comes from the water consumed directly as drinking and cleaning water for the animals. The category 'ancillary input housing' moreover also includes impacts due to use of Diesel fuel and electricity. The impacts of the feed are mostly due to the use of mineral fertilizers. The total volumetric amount of consumptive water per kg of FPCM is 5.0 to 5.5 L in our study; the result of the impact assessment is 1.5 to 1.6 L-eq. per kg FPCM. The amount of water used for drinking / cleaning is 4.0 L in the present system.

The rank order of the two systems appears robust as the "previous diet" product system (with the higher impact) might actually include irrigation in crop production of some feed ingredients purchased from suppliers.

Land Use: System 'current diet' has a 34% higher land use impact compared to system 'previous diet'. Main contributions are due to the use of arable land for feed production at the Grignon farm for the dairy cows as well as the older heifers and non-productive cows (maize -silage and grains-, alfalfa, pasture and oilseed rape). By contrast, sugar beet pulp, which makes up a larger fraction in the previous diet, has a comparably low profile in terms of land occupation. The rather low land occupation of beet pulp is due to being a co-product of sugar production of sugar beet. We applied an allocation factor of 3.8% for beet pulp (91.7% are allocated to sugar and 4.5% to molasses).

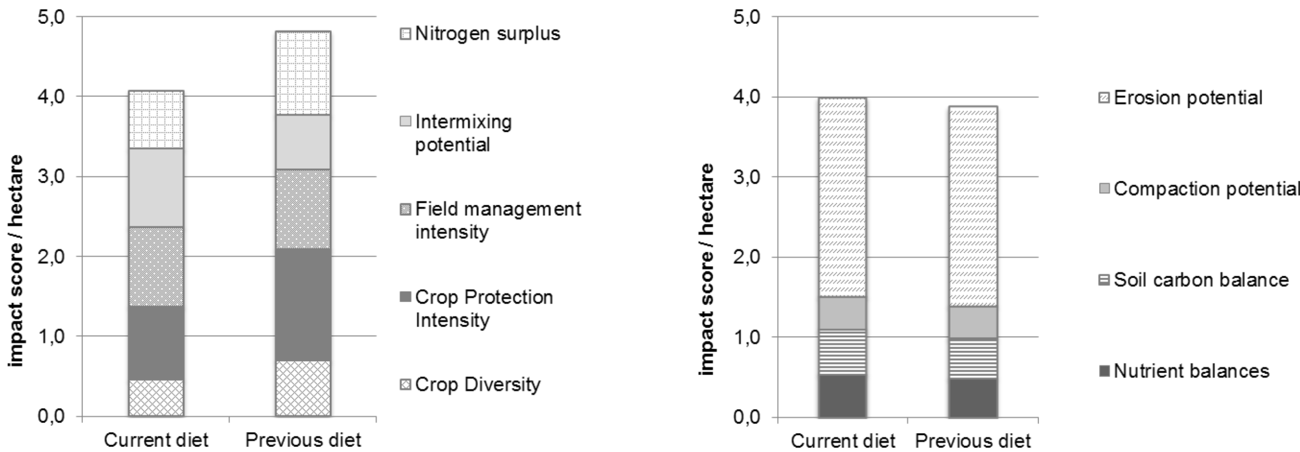
We also assessed other environmental impacts. The results for abiotic resource depletion and marine eutrophication likewise show an improvement for the current diet system relative to the previous diet system. Some other impact indicators show insignificant differences for the two systems, e.g. eco-toxicity potential, acidification potential and photochemical ozone creation potential. The fresh water eutrophication potential shows a worse performance of the current diet system. However, this result is strongly influenced by the contribution from avoided phosphate mineral fertilizer production.

3.2. Farm specific Biodiversity and Soil Assessment

Evaluation of biodiversity and soil indicators was carried out specifically for the agricultural area at the Grignon farm dedicated to the forage crop production (corn, alfalfa, oilseed rape, grassland and pasture, wheat). Wheat is included because wheat straw serves as an input for the cow bed and also as a minor feed component. Other areas, which are not associated with the fodder crops, were not considered.

Biodiversity (figure 3): Except for the indicator intermixing potential, system 'current diet' performs equal or better concerning all indicators. The previous diet performs better concerning the intermixing potential due to the inclusion of oilseed rape as a forage crop. Oilseed rape has a higher intermixing potential with the natural vegetation compared to the other crops. The indicators performing better for the current diet are the nitrogen surplus, the field management intensity, crop protection intensity and crop diversity. The indicator conservation area is determined by location and remains unchanged between both systems (not shown). Agri-environmental schemes were not implemented on the farm.

The evaluation of management indicators related to soil showed only minor differences between both systems.



Biodiversity　　Soil quality and conservation

Figure 3. Weighted impacts on biodiversity in agricultural areas (left) and soil quality and conservation (right) of the current and previous diet on agricultural areas for forage crop production at the Grignon farm.

3.3. Socio-Economic Impact Assessment

In this comparative assessment, the basic socio-economic figures of the farm remained were assumed to remain unchanged, to focus the comparison on the effects of the change in feed composition. The resulting impacts – both positive and negative indicators – attributed to system 'current diet' are generally higher at the Grignon farm, and lower for the farm-external activities. This is related to the higher amount of feedstuff produced on farm in system 'current diet' compared to system 'previous diet', where more feed was bought from the market. Hence Grignon farmers/farm employees, and local community potentially benefit from the new feed system in so far as potentially more labor activities are carried out on the farm itself, meaning that e.g., more wages, employment opportunities, and social security provisions are attributed to the milk production. On the other hand, the need for more land to produce the feed poses additional needs for productive area, with consequences for the land market and potential future restrictions in access to land.

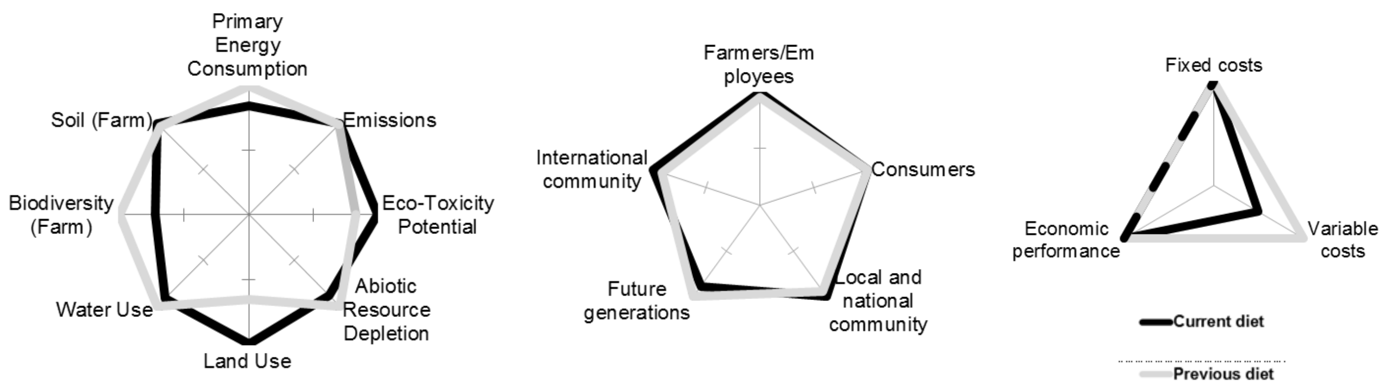
The economic assessment showed that the new feed (current diet) can potentially contribute to cost reductions for the farm. Direct costs to produce the higher amount of feed on the farm in the current diet are lower than the purchasing costs for the substituted components of the previous diet. The overall costs for use of fixed assets and total farm profits are unchanged in the two systems.

3.4. Sensitivity Analysis

Sensitivity analyses on multiple relevant methodological choices (allocation factors, milk productivity, emission factors) showed that allocation factors chosen for by-products (e.g. sugar beet pulp, distillers grain or rape meal), milk-meat allocation factor, milk productivity as well as the methane conversion factor for the manure management system have a significant influence on the absolute indicator results. The absolute deviations in GWP amounted to 20%, the standard deviation is 10%. The relative ranking of the alternative systems is however hardly affected, therefore, the conclusions drawn above remain valid.

3.4. Aggregated Results and Trade-off Analysis

Normalized and aggregated results are shown in Figure 4 to give a concise representation of the individual indicator results discussed in the preceding sections.



Environmental fingerprint | Social fingerprint | Economic fingerprint

Figure 4. Relative results obtained by internal normalization to the highest impact. Smaller figures indicate better performance. Dark line: system 'current diet'; grey line: system 'previous diet'. Social indicators for the stakeholder group 'Consumers' have not been evaluated in the present study.

The environmental fingerprint shows advantages of the new feed composition in the current diet related to a lower cumulative energy demand, abiotic resource depletion and the biodiversity potential of the agricultural area at the farm. The global warming potential is considered as part of the aggregated category emissions, and is lower for the current diet. The category emissions further contains the impact categories: acidification potential (AP), photochemical ozone creation potential (POCP), ozone depletion potential (ODP), and impact on water quality. The previous diet (old feed mix) has advantages related to a lower land use. The difference in eco-toxicity potential falls within the uncertainty of the USEtox impact assessment methodology. The higher impacts from land use in the current diet primarily are due to a higher land occupation on the farm itself. As shown in Figure 3, the biodiversity indicators related to on site management of the production area at the Grignon farm indicate an improvement (indicator 'Specific Impacts on Biodiversity in Agricultural Areas') with the current diet, and therefore the higher land use is balanced by a higher biodiversity potential on the farm area.

The social fingerprint shows that at the level of stakeholder categories, the impacts aggregated over the life-cycle lead to negligible differences between the two alternative systems.

4. Conclusions and Recommendations

The main objective of the farm is to reduce the climate change impacts and energy consumption of its produce. The present results clearly show that this objective was met with the introduction of the new cattle feed (current diet). Dairy cow feed crops produced at the Grignon farm contribute to lower GWP impacts of the current diet. However, the absolute GWP impacts foremost are due to emissions from livestock, which contribute with more than two-thirds to overall emissions. This underlines the relevance of investigating mitigation options of GWP impacts from livestock emissions. Mitigation strategies for CH_4 from enteric fermentation could be based on optimizing the feed mix for higher digestibility and usage of fat sources with reduction potential concerning methanogen bacteria. Regarding the manure management system, storage time and storing conditions (e.g. type of cover, temperature) are important parameters for reducing methane emissions. Furthermore, biogas plants provide excellent options for reduction of these emissions. Gastight covered biogas plants usually emit about 1 % of the CH_4 forming potential. In contrast to this, the liquid manure management system considered in this study leads to emissions in the dimension of 20% of the CH_4 forming potential. Additionally the energy produced by the biogas plant can substitute fossil fuels.

Few trade-offs between environmental consequences of introducing the new feed composition exist. The higher land use associated with the current diet is – at least to same extent – balanced by a higher biodiversity potential on the farm area. This may be further supported on the farm through targeted implementation of agri-environmental schemes, for example, flowering strips. The higher land use in system 'current diet' could also lead to a higher risk for indirect land use change. This has not been considered in the present study due to high uncertainty associated with such evaluations. Nevertheless this could be assessed in a further step for a better understanding of global environmental consequences of the two systems.

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Life Cycle Assessment of apples at a country level: the case study of Italy

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ABSTRACT

Assomela, the Italian Association of apple Producers Organizations (POs), has financed the Life Cycle Assessment (LCA) of the average apple produced in Italy. A study was carried out with the aim of providing an EPD certification and understanding the contribution of the different production phases to the carbon (C) footprint. For the cultivation phase data from several farms were sampled taking into account eight geographical areas and four main apple varieties cultivated in Italy. Data about the apples storage, processing, packaging and distribution phases were then collected from the POs. Four impact categories were considered: Global Warming Potential (GWP), Photochemical Ozone Creation Potential (POCP), Acidification Potential (AP) and Eutrophication Potential (AP).Results show that 1 kg of apple, in a cradle to retailer perspective, have a GWP of 0.20 kg CO_2 eq., a POCP of 0.18 g C_2H_4-eq., a AP of 1.12 g SO_2-eq and a EP of 0.62 g PO_4^{3-}-eq. The main contributor of the C footprint during the cultivation phase was the consumption of fuel for machinery, which significantly changed according to the distance from the farm center and the field size.

Keywords: Environmental product declaration, carbon footprint, fuel consumption

1. Introduction

Assomela, the Italian Association of apple Producers' Organizations (POs) representing 80% of the national production, promoted a project for the certification of the environmental impact of apple production in Italy. Assomela brings together VOG (Marlene), VI.P and VOG-Products in the province of Bolzano, Melinda and La Trentina in the province of Trento, COZ and Nord Est in the region Veneto, Melapiù in the region Emilia Romagna, Rivoira and Lagnasco in the region Piemonte and Melavì in the region Lombardia (Table 1).

Table 1. Main characteristics of the Producers' Organization (POs) involved in the study

POs	Head office	Apple production (t year^{-1})	Cultivated land (ha)
La Trentina	Trento	100,000	1,800
Melinda	Cles	330,000	6,500
VI.P	Laces	330,000	5,100
VOG	Terlano	660,000	10,600
Lagnasco	Lagnasco	15,000	348
Melavì	Ponte in Valtellina	26,000	NA
OP NordEst	Verona	15,000	400
Rivoira	Verzuolo	30,000	600

Several varieties are cultivated in the area under study but for the LCA only the main four (that together represent the 76% of the total production) were considered (table 2).

Table 2. Varieties cultivated by the 8 Producers Organizations (POs)

Variety	Percentage on the total production
Golden Delicious	44%
Gala	14%
Red Delicious	11%
Fuji	7%
Granny Smith	5%
Other varieties	19%

Objectives of the study were 1) to assess the environmental impacts of the apple production chain in Italy following the *Product Category Rules* for *Fruits and nuts* of the International EPD System and 2) to characterize the processes occurring during the cultivation phase that most affect the carbon footprint of the main apple variety.

2. Methods

2.1. Functional unit

The functional unit of the study is 1 kg of apples intended for fresh consumption, delivered at the retailer.

2.2. System boundaries

The considered system includes all the activities carried for the agricultural production, the storage, the processing and the distribution. According to the PCR the nursery phase has not been considered since the average orchard duration may exceed 25 years and thus the impacts of this phase can be considered negligible.

The main factors considered for the field phase are the use of oil, water, pesticides and fertilizers (see also table 4).

After the harvest, apples are stored in controlled atmosphere (where they stay in cool chambers at low oxygen concentration), until they are requested by the market. Then they are packed and finally delivered.

The distribution of apples occurs both in Italy and abroad, mainly through land transport by truck even if a small percentage is shipped by sea.

2.3. Treatment of data

The informations presented refer to eight of the ten Producer Organizations associated to Assomela, operating in the regions of Trentino Alto Adige, Piemonte, Lombardia and Veneto. Since the goal of this declaration is to provide information typical of the whole association, the data have been processed in a way that allows to create different averages between the POs participating to the project, using weighting factors based on production volumes. In detail, the average has been organized in three different levels:

- (M1): the average for each variety cultivated by each PO;
- (M2): the overall Assomela average for each variety, calculated by using the single variety production quantity of each PO as weighing element;
- (M3): the overall average for all the apples produced by the POs associated to Assomela.

2.4. Main assumption of the LCA approach in the four production phases

Cultivation

Consumption of water and diesel oil was estimated according to the real consumption of the selected farms. Data on other consumption (fertilizers and pesticides) have been obtained by the production specifications of the areas interested and then validated with specific information.

Data about yields were calculated considering the average age of the trees and the production volume. Data about produced waste were collected by APOT, the fruit and vegetable organization of Trentino. The land use change was not included in the calculation, since almost all the orchards are in the studied areas last for more than 20 years.

The global warming potential caused by the cultivation phase was additionally assessed by considering an average “Golden delicious” apple orchard managed according to the integrated fruit production guidelines of South Tyrol, yielding 62 t ha^{-1} y^{-1} (Mazzetto et al., 2012).

To investigate the role of the farm structure on the carbon footprint of apples, simulations about fuel consumption in scenarios with different field sizes and field distances from the farm center were carried out. A farm with a total surface of 3.9 ha, entirely cultivated with apple trees, representative of many farms in Trentino Alto-Adige, was selected as model for the simulations. The total farm surface was split into 15 fields with an average size of 0.26 ha and a distance from the farm center ranging from 200 to 1200 m.

Processing

In this phase electric energy consumptions, water consumption and waste production both for processing and for storage have been considered.

Data were collected from a sample of plants.

Packaging

The presented data refer to the selling apples considering the use of 1 plastic bag for 1 kg of apples.

Distribution

The impacts referred to the transportation phase have been calculated considering an average transport of 850 km by truck and 250 km by ship, which takes into account the Italian and European markets, as well as smaller amounts distributed overseas, in the Asian and North African markets.

3. Results and discussion

The main results of the LCA approach are shown in Table 3 and Figure 2.

Table 3. Environmental impacts of the apple production chain in the four considered impact categories. Data are referred to 1 kg of apples commercialized in a plastic bag.

Potential impact	**Cultivation**	**Processing**	**Packaging production**	**Transportation**	**Total**	
Global Warming Potential	0.04	0.06	0.01	0.09	0.20	kg CO_2 eq
Photochemical Ozone Creation Potential	0.06	0.02	0.03	0.07	0.18	g C_2H_4
Acidification Potential	0.31	0.22	0.04	0.55	1.12	g SO_2 eq
Eutrophication Potential	0.41	0.08	0.01	0.12	0.62	g PO_4-eq

Transportation had the highest impact on the Global Warming Potential, followed by processing, cultivation and packaging production (Figure 1).

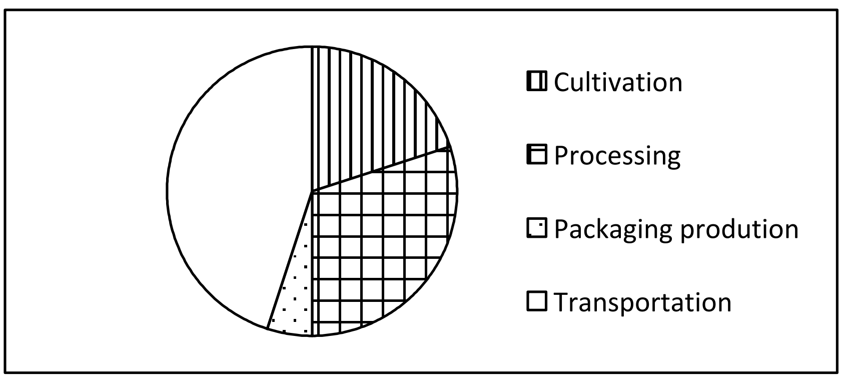

Figure 1. Contribution of the different production phases to the Global Warming Potential impact category.

Transportation has a high GWP because it occurs mainly by trucks. Transport by sea or railway could be a solution to decrease these emissions (Blanke and Burdik, 2005). Processing has also a high relevance because in this phase cool storage is included and it is a process that requires a significant amount of electric energy.

Cultivation is the third contributor in terms of GWP. The main impacts in this phase are due to the amount of diesel for machinery and to production and use of fertilizers (table 4)

Table 4. Carbon footprint of the Golden delicious apple during the cultivation (cradle to gate) phase (data from Mazzetto et al., 2012).

Source of emission		CO_2 emissions (GWP)	
		kg CO_2 eq kg^{-1}	%
Infrastructure	Buildings	0.002	4
	Machinery	0.013	25
	Planting scaffold	0.001	2
	Irrigating system	0.007	14
	Plant material	0.001	2
Annual management	Fertilizers	0.006	12
	Herbicides	0.002	4
	Pesticides	0.003	6
	Labour	0.001	2
	Fuel	0.015	29
TOTAL		0.051	100

As reported by Zanotelli et al.(2014), the total value of C footprint for the cultivation phase of apple is relatively low when compared with other fruit cultivation (data expressed in MJ kg^{-1} of primary energy use) also due to the high yields reached by modern apple orchard that allow a good use efficiency of external resources. Fuel consumption resulted to be among the most important impact factors also in apple produced in other countries, and in other fruit production chain like olive and grape (Zanotelli et al., 2014).

The sensitivity analysis carried out for the cultivation phase with the analytical approach, quantifies the linear increase of fuel consumption observed by increasing the distance of the fields from the farm center (Table 5). The consumption due to the machines transfer can reach 50% of the total when the fields (with an average size of 0.26 ha) are located at more than 1500 m from the farm (Tab 5). The simulation reported in table 5 has been carried out considering an average apple production of 60 t ha^{-1}.

Table 5. Simulation of changes in fuel consumption for machinery (as total yearly amount and as % due to the transfer of machinery) due to the increasing distance of the apple orchard from the farm-center.

Distance farm center -field	Fuel consumption	% of fuel consumption due to the transfer of machinery
m	kg ha^{-1}	%
200	294	10
500	341	26
1000	426	36
1500	503	45
2000	581	52
4000	906	67

A second simulation aimed to assess the variation in fuel consumption when the apple fields, 1000 m far from the farm center and yielding 60 t ha^{-1} of apples, varies their size from 0.26 to 8 ha. Results show that there is a sharp decline in fuel consumption at increasing field size (Tab. 6).

Table 6. Simulation of changes in fuel consumption for machinery (as total yearly amount and as % due to the transfer of machinery) due to the increasing size of the fields. An average yield production of 60 t ha^{-1} and a distance of the fields to the farm center of 1000 m was considered in the sensitivity analysis.

Field surface	Fuel consumption	% of fuel consumption due to the transfer of machinery
ha	kg ha^{-1}	%
0.25	426	36
0.5	358	30
1	306	23
2	277	19
4	261	17
8	254	17

4. Conclusions

Considering the difficulties in performing an LCA study at a country level, especially due to the adoption of a good sampling strategy, we believe that the approach used in this study reconciles feasibility of data collection and statistical significance.

The carbon footprint of the whole apple production chain is dominated by the emission occurring in the transport phase from the storehouse to the final market. The use of more efficient transportation means with respect to truck should be explored in the future.

The good level of apple yield per hectare reached by modern orchard (approx. 60 t ha^{-1}) indicated a good use efficiency of external resources which is reflected in a relatively low C footprint of the apple cultivation phase (0.04 - 0.05 kg CO_2-eq). Anyway a better farm organization in terms of field distance from the farm center and average field size could further diminish the current C footprint value.

This analysis contributed to increase the awareness of environmental impacts of apple production and constitute a scientific basis for improving the sustainability of the whole apple production chain.

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Impact of Sustainability Labeling in purchase intention and quality perception of dark chocolate

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ABSTRACT
Labeling is an important tool for consumer's perception of sustainability and quality of a product. Focusing on measuring the impact of sustainability labeling (seal and/or indication of Organic, Origin and Quality, and Sustainable Agriculture) in purchase intention and quality perception of products labeled by the quality and sustainability criteria, this study aimed to investigate dark chocolate (pack-1 kg) with six different percentage of cocoa, with and without such labeling. A blind test was carried out in the first evaluation session, in which the samples 1, 2 and 6 were better accepted by consumers. In the second session, all judges were able to see the percentage of cocoa and the label of each sample, and the chocolates 1 and 2 obtained the highest acceptance scores. It is possible to conclude that labeling had a positive impact on purchase intention and quality perception of dark chocolates.

Keywords: Sensory analysis, Purchase intention, Dark chocolate, Labeling, Sustainability

1. Introduction

Cacao cultivation is an important agricultural activity in areas of hot-humid climates. In 2009/2010, according to the International Cocoa Organization (ICCO, 2010), Brazil was the sixth largest producer of cocoa in the world and occupied the fifth position on processing cocoa for obtaining the main derivatives used the chocolate industry (liquor / cocoa mass and cocoa butter).

Brazil is the third largest producer of chocolate in the world; in 2012 the chocolate production has increased by 3.1% over 2011, with 732 000 tons. The country remains the fourth largest consumer of chocolate in the whole world, with per capita consumption of 2.2 kg per year, showing an optimistic scenario for the coming years, since three years ago this number corresponded to 1.65 kg. According to a balance sheet released by the Brazilian Association of the Chocolate, Cocoa, Peanut, Candy and Derivatives (ABICAB), there was also an increase in apparent consumption of 3.7%, with 717 000 tons (ABICAB, 2014).

These data evidence the economic and social importance of cacao cultivation. Brazil is the only country with the complete production chain, with a significant production, cacao processing, and high consumption of chocolates.

The sectors of cocoa cultivation and chocolate production have developed several projects related to sustainability around the world. Two certifications have focused on sustainability in the development of cocoa / chocolate: as organic chocolate and products with designation of origin, for example with Rainforest Alliance Certified seal. In Europe and in the United States, there is an increase in the organic sector, thus companies have been establishing some guidelines for its manufacture. According to the Organic Trade Association, consumers buy organic products not only for their nutritional benefits, but also to avoid the consumption of pesticides and additives, in addition to maintaining the environment and promote sustainability. This customer profile is very concerned about the social and environmental aspects of food, and possibly will keep brand loyalty if the product provides positive social and environmental impacts, despite the costs may be higher. Origin cocoa is a trend in the chocolate segment, since the characteristics of each region, such as climate and soil conditions influence the cocoa quality and can provide good results on cocoa liquor and chocolate flavor. As with wine and coffee, the place of origin of cocoa production is taken into account due to the different flavor notes peculiar to the region or farm that cultivates the fruit. In addition, proper farming, environmentally correct handling and processing allow obtaining better quality products, which also promotes sustainable regional development. The concept of origin cocoa considers not only the place of production, but also the environmentally management, with a focus on fruit quality and social concern for producers (SADAHIRA, 2010).

The role project aims to apply the Ecodesign (ISO/TR 14062, 2002) methodology in the production chain of cocoa in Brazil. The objective is evaluate the impact of sustainability labeling to contribute for the development of chocolates with focus on sustainability, in a Brazilian industry that export to over 30 countries and is one of the topping leading manufacturers in Latin America (HARALD, 2014). In the scope of the project are chocolates with high cocoa content (>50%), business-to-business packs (>1kg), from "cradle to grave" LCA (ISO 14040, 2006; ISO 14044, 2006) (cocoa and sugar producer, packing, chocolate manufacturing and consumption). A research data collection at supermarkets identified two kinds of certification focused on sustainability in the development of cocoa/chocolate: certification of agricultural practices as organic cocoa and cocoa with designation of origin (without any seal or with Rainforest Alliance seal). The chocolates were produced and sensory evaluated by 126 consumers without restriction of age or sex.

This paper, a part of Ecodesign Project, aims to characterize by sensory analysis the profile of Brazilian consumers as the impact of labels and certification of organic, origin and quality or sustainable agriculture (Rainforest) on the acceptance and purchase intention of chocolates.

2. Methods

2.1. Manufacturing process

A research data collection was carried out at different supermarkets in the city of Campinas-SP to identify the organic chocolates and those with designation of origin or with Rainforest Alliance Certified Seal, from national manufacturers. Then, six dark chocolates containing different percentages of cocoa were selected for molding and demolding steps (Table 1).

Table 1: Chocolate samples selected in the Brazilian market, with label or indication of organic, with designation of origin and quality or with Rainforest Alliance Certified seal.

Samples (Chocolates)					
1*	2*	3*	4*	5**	6**
53% Origin and *Rainforest*	63% Origin and Quality	70% Organic from Amazon	75% Organic from Bahia	70% cocoa	45% Dark

Samples 1: 53% origin cocoa and with Rainforest Alliance Certified seal; 2: 63% origin cocoa with quality indication "*Salon du Chocolat* Award Winner"; 3: 70% organic cocoa from Amazon-Brazil; 4: 75% organic cocoa from Bahia-Brazil; 5: 70% cocoa; 6: 45% dark chocolate; *seal and/or indication of Organic, Origin and Quality or Sustainable Agriculture (*Rainforest*); ** without seal and/or indication of Organic, Origin and Quality, or Sustainable Agriculture.

The six chocolate samples were melted and submitted to tempering, molding, cooling and packaging. Then, the samples were melted in a microwave and subjected to the tempering step performed manually on a marble table. The chocolate mass was warmed to 50 ± 1.0 ° C and then cooled to 29 ± 1.0 ° C under constant movement at a rate of 2 °C / min. The pre-crystallization was monitored by a temper meter considering the tempering index between 4.0 and 6.0. The chocolate was dispensed manually into preheated polypropylene rectangle-shaped molds. Then, the molds were subjected to vibration to remove air bubbles, and the samples were cooled in cooling tunnel. The chocolates were demolded, packaged in aluminum foil and stored in a chamber at 18 °C.

2.2. Sensory Analysis

The impact of labeling on the acceptance and purchase intention of consumers was carried out with 126 consumers of dark chocolate, in the Sensory Analysis Lab, Department of Food Technology (DTA / FEA/UNICAMP) and Laboratory of Sensory Analysis of Cereal & Chocolate Research Center (Food Technology Institute - ITAL). The chocolates were evaluated by consumer acceptance test with consumers of dark chocolate (STONE & SIDEL, 2004).

The sensory evaluation of chocolates was performed according to a randomized complete block design, with no restrictions on age, gender and social class. The test was divided into two sessions, with a blind test performed in the first session, without revealing the origin or the certification seals.

In the second session, after the blind test, the assessors were informed about Organic and *Rainforest* seals, certificates of origin, and then the origin and/or certification of each chocolate sample was revealed, followed by the purchase intention and sensory acceptance analysis.

In both sessions, the samples were evaluated for the attributes aroma and chocolate flavor, chocolate melting in the mouth, bitterness, acidity, hardness or force required to break the chocolate, and overall impression, through a 9-point scale anchored as follows: 9 "like extremely"; 8 "liked"; 7 "liked moderately"; 6 "liked slightly"; 5 "neither liked nor disliked"; 4 "dislike somewhat"; 3 "dislike moderately"; 2 "dislike very much"; and 1 "dislike extremely". The purchase intent was assessed using a 5-point scale: 1 "definitely would buy"; 2 "probably buy"; 3 "I doubt if I would buy"; 4 "probably would not buy"; and 5 " definitely would not buy." The data were subjected to analysis of variance (ANOVA) and means were compared by Tukey's test at 5% significance level.

3. Results

Table 2 presents the results of the blind acceptance test in the first session with 126 consumers.

Table 2. Blind acceptance test of six different samples

Chocolate samples	Attributes						
	Flavor	Bitterness	Acidity	Aroma	Hardness	Melting	Overall impression
1 53% Origin and *Rainforest*	6.64 a	5.97 a	5.38 ab	5.98 a	6.12 ab	6.71 a	6.65 a
2 63% Origin and Quality	6.38 ab	5.86 a	4.96 ab	6.12 a	6.06 ab	6.39 ab	6.21 ab
3 70% Organic from Amazon	5.88 bc	5.89 a	4.75 b	6.22 a	6.07 ab	5.95 b	5.73 bc
4 75% Organic from Bahia	2.96 d	3.87 b	2.93 c	4.79 b	5.14 c	4.75 d	2.87 d
5 70% cocoa	5.53 c	5.24 a	5.12 ab	6.11 a	5.66 bc	5.36 c	5.60 c
6 45% Dark	6.15 abc	5.27 a	5.58 a	6.01 a	6.28 a	6.54 a	6.32 ab

Averages in the same column followed by the same letter are not significantly different by Tukey test ($p \geq 0.05$).

Although the sample 1 was more accepted by consumers, with a mean score lying in the category "moderately liked" for the attribute overall impression, it was not significant ($p \leq 0.05$) different from the samples 2 and 6, with a mean score lying in "slightly liked ".

With respect to the attribute chocolate flavor, sample 1 had the highest acceptance score in the category "moderately like", but did not differ ($p \leq 0.05$) from the samples 2 and 6, with a mean score lying in "slightly liked".

Concerning the bitter taste, despite the samples 1, 3, and 2 had the highest scores in the category "slightly liked", it did not differ statistically from the samples 5 and 6, which scored in "neither liked nor disliked".

As the attribute acidity, sample 6 was the most widely accepted, lying in the category " slightly liked ", but did not differ at a significance level of 5% from the samples 1, 2 and 5 that scored in "neither liked nor disliked".

The low acceptance of the sample 4 when compared to the other samples is mainly due to the attributes flavor, bitterness, acidity, aroma and melting which have received lower mean scores and influenced the overall impression. The results also indicate that Brazilian consumers enjoy chocolate containing low levels of cocoa than chocolates with high cocoa content.

Table 3 presents the purchase intention in the blind test.

Table 3. Purchase intention of chocolates in the blind test.

Purchase intention	Samples (blind test)					
	1 53% Origin and *Rainforest*	2 63% Origin and Quality	3 70% Organic from Amazon	4 75% Organic from Bahia	5 70% cocoa	6 45% Dark
Definitely would buy	41.27%	36.51%	26.19%	3.97%	26.19%	39.68%
Probably buy	42.06%	36.51%	25.40%	3.97%	28.57%	29.37%
I doubt if I would buy	9.52%	16.67%	29.37%	13.49%	22.22%	19.84%
Probably would not buy	5.56%	8.73%	12.70%	34.92%	18.25%	7.14%
Definitely would not buy	1.59%	1.59%	6.35%	43.65%	4.76%	3.97%

In general, samples 1, 2 and 6 had the highest scores in the purchase intention test. In contrast, sample 4 had the highest scores in the category "certainly would not buy", with over 40% intentions, demonstrating that the flavor has influenced purchase intention in the blind test.

Table 4 shows the results of the acceptance test after the information of the samples with or without labeling.

Table 4. Acceptance test of the samples with or without labeling

Samples	Attributes (with or without labeling)						
	Flavor	Bitterness	Acidity	Aroma	Hardness	Melting	Overall impression
1 53% Origin and *Rainforest*	7.01 a	6.37 a	5.62 a	6.55 ab	6.41 a	6.84 a	6.87 a
2 63% Origin and quality	6.85 ab	6.27 a	5.56 a	6.67 a	6.49 a	6.75 ab	6.90 a
3 70% Organic from Amazon	6.13 c	6.03 ab	4.84 b	6.37 abc	6.26 ab	6.30 b	6.11 bc
4 75% Organic from Bahia	3.20 d	3.96 c	3.02 c	5.01 d	5.32 c	5.02 d	3.03 d
5 70% cocoa	5.71 c	5.53 b	5.34 ab	6.09 bc	5.80 bc	5.60 c	5.60 c
6 45% Dark	6.30 bc	5.50 b	5.65 a	5.94 c	6.40 a	6.55 ab	6.28 b

Averages in the same column followed by the same letter are not significantly different by Tukey test ($p \geq 0.05$).

In the second session, the samples containing the seals were presented to the assessors. Both samples 2 and 1 were the most accepted by consumers, with a mean score lying in "like moderately" for the overall impression, with no significant differences at 5% level.

The blind test showed that chocolate 1, 2, and 6 were the most accepted for the majority of the attributes. However, in the second session where the assessors knew the cocoa content and the seals of each sample, samples 2 and 1 were the most accepted and received higher scores than those in the blind test, showing that the labeling with seals had an impact on the overall acceptance of chocolates.

For the attribute flavor, the sample 1 had the highest score lying in the category "like moderately", followed by the sample 2 with a mean score also in the category "like moderately".

Samples 1, 2 and 3 had the highest scores for the attribute bitter taste in the category "liked slightly", and differed statistically from the samples 5 and 6.

Again, the sample 4 was less accepted by consumers for flavor, bitterness, acidity, aroma and melting attributes at a significance level of 5%, with a mean score on the overall impression lying in the category "dislike moderately".

Even after the information of the organic seal, the chocolate 4 remained less accepted than the other chocolates, demonstrating that the taste was the most important factor for consumers.

As shown in Table 4, after the revelation of the seals, the acceptance scores increased for samples 1, 2 and 3. The sample 2 contained an origin cocoa with quality flavor indication, as twice winning the award for best chocolate flavor at the *Salon du Chocolat* in Paris, France. The sample 1 contained the sustainable agriculture *Rainforest* seal, and the sample 3 contained the declaration of organic cocoa from Amazon-Brazil.

It is verified in the second session that the samples were separated into groups, once the samples 2 and 1 received the highest acceptance scores for almost all attributes, and did not differ at a 5% significance level. On the other hand, samples 3, 5 and 6 formed the second group receiving the highest acceptance scores, and did not differ at a significance level of 5% for the attributes evaluated.

In general, it is possible to say that the labeling influenced the sensory acceptance of chocolates after the second session, when the assessors knew the percentage of cocoa and the seals of each sample. Samples 1, 2 and 3 received the highest scores from one test to another, and no changes were observed for the sample 5, evidencing the positive impact of labeling.

Table 5 presents the results of consumer purchase intent after the information about labeling of chocolates.

Table 5. Purchase intention in the second session after information about labeling.

Purchase intention	Samples					
	1 53% Origin *Rainforest*	2 63% Origin and quality	3 70% Organic from Amazon	4 75% Organic from Bahia	5 70% cocoa	6 45% Dark
Definitely would buy	51.6%	50.0%	29.4%	6.3%	24.6%	42.9%
Probably buy	36.5%	33.3%	35.7%	7.1%	34.1	27.0%
I doubt if I would buy	9.5%	12.7%	19.0%	18.3%	20.6%	19.8%
Probably would not buy	2.4%	3.2%	11.9%	26.2%	14.3%	7.9%
Definitely would not buy	0.0%	0.8%	4.0%	42.1%	6.3%	2.4%

Again, the chocolate 1, when compared with the blind test, obtained the largest percentage of purchase intention with 51.6%, followed by samples 2 and 6.

It is noticed that after revealing the information of the samples to consumers, the positive effect on purchase intention increased and the negative effect decreased, except for the sample 5. Regarding the sample 4, the uncertainty regarding the purchase intention increased, thus showing the seal and / or indications may have impacts on the sensory acceptance and purchase intention by consumers.

4. Discussion

Johansson et al. (1999) found an impact of labeling on the form of production (conventional and organic) in the sensory preference of tomato consumers. After informing about the cultivation techniques, the authors observed that the labeled organic samples increased the preference scores, but that information was less important when the tomatoes were sweeter and had more intense taste when compared with those grown ecologically and presenting high acid taste.

Levin and Gaeth (1988) studied the effect of labeling on the perception of four sensory attributes of meat. The judges were informed that a sample contained "75% lean ground beef" and the other contained "25% fat ground beef". Although both samples contained the same fat content, information was passed differently for the judges. The results showed that the 75% lean beef was evaluated as low fat with better quality than the sample labeled 25% fat beef.

In this study, it was observed that the labeling had an impact on the consumer acceptance and purchase intention, once those samples containing quality seals presented the highest sensory scores when compared with those chocolates without such labels.

5. Conclusion

It was possible to observe that the consumer's behavior changed when quality and sustainability labels have been informed, since the sensory acceptance and purchase intention increased for the samples containing the seals. Moreover, it appears that the sensory attributes were also important for Brazilian consumers, once among the samples with seals, the chocolates with lower cocoa content were more accepted in both sessions.

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Paths to a sustainable food sector guided by LCA – exemplified by pork production

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ABSTRACT

To describe a more sustainable food sector, a supply chain approach is needed. Changing supply chains inevitably means that a range of attributes of the product and its system will change. This project will take on this challenge and deliver detailed descriptions of supply chains of six commodities from a Swedish region in 2012; Milk, cheese, beef, pork, chicken and bread. The set-up of the project was that experts on production along the supply chain design environmentally improved systems. The next step was to challenge the improvements considering their possible consequences on products and systems from different perspectives: food safety, sensory qualities, animal welfare, and consumer appreciation and (only for primary production) costs. The final supply chains were quantified by life cycle assessment (LCA), and they were again assessed from the perspectives mentioned above. Results will be generated during August 2014 and comprise both single-product LCA and region-wide impacts of the future scenarios.

Keywords: Sustainable food chains, Food system scenarios

1. Introduction

To design a more sustainable food production we need to look at the whole supply chain, from the primary production and its inputs, to the consumers. This is obvious for the LCA community and is being embraced by a growing part of the society. LCA research and case studies are critical for understanding the impact profile of products and systems. Most studies aiming at proposing and analyzing food production systems with improved sustainability performance use an aggregated approach for identifying improvement potentials in different steps of the life cycle, where improvement potentials are based on general assumptions on possibilities in each step separately. This is a useful approach to identify and quantify the possibilities and may work as a "semi back-casting exercise" as it provides a more realistic future state that can be used as a starting point for focusing research and development on how to implement improvements. However, such an approach is less valuable for practical information on relevant actions needed by food chain actors (farmers, industry, retailers, consumers and various public decision-makers).

To move LCA research closer to more hands-on decision making, novel approaches are needed. In order to design environmentally improved supply chains on a more detailed level than what is usually accomplished by LCA there is a need for new working methods that combine the systems perspective from LCA with deeper and more detailed knowledge about production systems.

Another extremely important aspect of food systems is the broad range of food related aspects, such as food safety, food quality and animal welfare. Consumer reactions are also critical when new solutions for the food supply chain are introduced. Furthermore, overall economic viability is a condition to be met. Changing food supply chains inevitably means that a range of these attributes of the product and its system will change, and neglecting this in studies will seriously hamper the interest from stakeholders and delay or stop necessary improvements in environmental performance.

The aim of the whole project was to present solutions for specific food supply chains with improved sustainability performance that are realistic and concrete enough for food chain stakeholders to use in their decision-making. The focus is on improved production in a short-term perspective; whereas changes in consumption patterns are beyond the scope of the project. A second aim was to develop and apply an integrated working process involving different disciplines and synthesizing their knowledge.

The objectives are to:

- define present and detailed descriptions of supply chains of six commodities from the county Västra Götaland in Sweden in 2012; Milk, cheese, beef, pork, chicken and bread
- describe future scenarios for these supply chains, producing the same amount of commodity as in 2012, where the environmental performance is improved while important attributes mentioned above (e.g. product safety and quality) are improved or kept at present levels
- quantify the environmental impact of these supply chains and compare them with the present situation as reference using LCA
- quantify the costs in primary production for scenarios and the reference, based on present cost levels.

This paper describes parts of the project, namely the supply chain for smoked, sliced ham, which includes animal production, feed production, industrial processing, packaging and distribution. Description will be presented to highlight the processes as well as solutions. During early autumn 2014 these solutions will be evaluated by LCA and presented at the conference.

2. Methods

The set-up of the project was that experts on production along the supply chain (agronomists, animal scientists, food engineers, packaging and supply chain management experts) designed a reference production system (in the following text referred to as "reference"), reflecting the present system. Three environmentally improved systems for each product chain were developed by the experts using literature and their own expertise combined with expertise on environmental impact of food supply chains. The close cooperation between different food chain experts and also with experts representing different "attributes" (food safety, animal welfare, food quality etc.) ensured overall improved supply chains, so called solution scenarios. To avoid trade-offs between environmental impacts within a particular system, the solution scenarios were designed in three versions in order to improve the performance of three "goal scenarios", corresponding to clusters of usually compatible environmental objectives for Swedish conditions. Finally the solution scenarios for the supply chains will be quantified using LCA and cost calculations (only agriculture). The general working process is described in Figure 1.

The time frame for implementation of improvements was five to ten years, so it basically meant gathering available knowledge to functioning supply chains. The primary production part included the entire animal- and/or crop production system within the specified region (beef, dairy, pig, chicken and bread grain production) while beyond primary production and primary processing (slaughter, milling) specific products were studied (smoked ham, frozen chicken fillets, sirloin steak, aged cheese, drinking milk and bread). The reason for limiting the later stages of the chain to just one single product is the existence of a huge diversity of products in later stages; a farmer delivers one product, the slaughter pig, while the meat processor delivers hundreds of products through a number of distribution channels. Of course the farming system is very complex, with numerous inputs and connections between parts, but the products are much fewer so it is possible to follow the total flow. This approach means that we will deliver LCA results for the entire primary production for the region studied, but only case study results for the chain beyond primary production. The retail and consumption stage is not included; the system boundary is set at the retail intake. The products were selected using criteria such as market volume and potential challenges to one or more attributes.

The approach also included interactions between the different agricultural systems. Hence, the crop production part of the chains, where each production system produced an important part of the feed required, and the manure management, with the intention to use the manure at the farm, functioned as the common space for this interaction. Feed can be, and often is, produced and traded between different farms. The total need for feed, quantified by animal nutrition experts aiming at high production efficiency and low environmental impact from feed raw materials corresponding to the different "goal scenarios", decided the design of crop rotations at the different types of farms. As a starting point, possible rations were assessed using emission factors from a feed database developed at SIK. In addition, wheat for bread production was also to be produced by the farm, in particular by specialized crop farms. Through discussions, the feed ration formulations were adjusted to better facilitate improvements in crop production, including manure management, in order to find compromises between optimal feeding and optimal crop production systems. Throughout the process the work was guided by food LCA experts that supported in decision-making, ensured the collection of relevant data in an appropriate format and facilitated discussions between expert groups.

The next step was to challenge the improvements considering their possible consequences on products and systems from different perspectives: food safety, sensory qualities, animal welfare, and consumer appreciation and costs for primary production. All scenarios were assessed, mainly qualitatively, and modified using an iterative process in order to take all perspectives into account at the same time in the final supply chains.

The finalized supply chains will be quantified by LCA and by cost calculations for primary production part of the chain. Furthermore, they will be assessed for possible consequences mentioned above.

The functional units for the primary production were the total production volumes for year 2012, one kg of product at farm gate, and, for the post-farm stage, one kg of packed product of the specific type chosen at the point of reception at retail. Results will be generated during Q3 and Q4 2014 and comprise both single-product LCA and region-wide environmental impacts and resource use for the future scenarios.

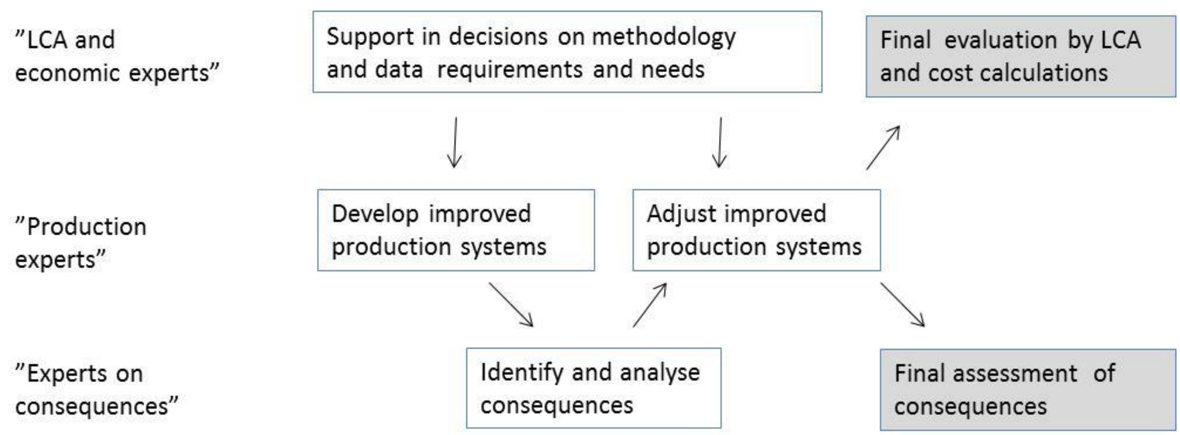

Figure 1. Principal description of the working process in the project, shaded boxes are not included in this article

3. Results

3.1. Goal scenarios

First, the generally most critical environmental impacts were identified using literature (Rockström et al., 2009, SEPA, 2013 (Sweden’s official environmental goals)). The next step was to “match” these environmental impacts with the full list of impact categories in LCA as defined in ReCiPe (Goodkoep et al., 2009) and identify impact categories where food production has an important role. This was made using published LCA and normalization. Based on this a priority list of impact categories was developed, where the highest priority was given to impacts where food systems have a large impact and the impact itself were identified as globally and nationally important (Table 1).

Table 1. List of Mid-point impact categories in ReCiPe, priority based on 1) global importance of category and 2) how important food systems are to total impacts (project internal assessment).

Impact category	Priority	Activity in the food sector contributing to the specific impact category
Marine eutrophication	1	Nitrogen turnover, use of phosphorous
Freshwater eutrophication	1	Nitrogen turnover, use of phosphorous, organic matter from soil
Freshwater ecotoxicity	1	Pesticides, Nitrogen turnover use of phosphorous, antibiotics, detergents in agriculture and industry
Biodiversity	1	Pesticides, agricultural practices general, crop rotations, management, Nitrogen turnover, use of phosphorous
Climate Change	1	Energy use, transports, nitrogen turnover, manure management, ruminants, soil, carbon flows, deforestation, refrigerants
Agricultural land occupation	1	Feed use, yields for crops
Minerals consumption	1	Use of phosphorous and sulphur, Metals used in machinery and equipment
Terrestrial acidification	1-2	Nitrogen turnover and manure (NH_3), transports, energy use, fertilizers
Terrestrial ecotoxicity	2	Pesticides, heavy metals e.g. in fertilizers
Natural land transformation	2	Feed use, yields for crops
Fossil fuel consumption	2	Production of chemical fertilizers, transports, energy use
Water consumption	2	Irrigation, cleaning (agriculture, industry)
Ozone depletion	2	Refrigerants, N_2O from nitrogen turnover
Human toxicity	2	Heavy metals, pesticides, antibiotics and natural toxins in foods
Photo-Chem. ozone formation	2	Transports, energy use, methane (from manure and ruminants)
Particulates	2-3	Transports, energy use
Urban land occupation	3	Infrastructure for industrial production, distribution and retail
Marine ecotoxicity	3	
Radiation	3	Electricity use (nuclear power)

A grouping was made where impact categories corresponding to environmental objectives that rarely are in conflict with each other were put in the same group. This was based on previous studies and experiences of food LCA. Furthermore impact categories corresponding to potentially conflicting goals were put in different groups (based on a large number of LCA studies). This resulted in three goal scenarios, presented in Table 2. The goal scenarios were then used as “compasses” for the development of the solution scenarios. Several measures were applicable in more than one solution scenario, but this kind of overlap is not a flaw to the method since they are accounted for only once in each solution scenario. The use of goal scenarios was a support for making decisions in cases there were measures that would lead to conflicts between environmental goals but still were interesting for one of the goals. For the post-farm chain it became evident that most improvement measures fell into scenario three. Hence only one solution scenario per product was developed for this part of the supply chain.

Table 2. Goal scenarios developed in the project

Goal scenario	Impact categories that will be addressed in solution scenarios	Examples of measures and connections to the production system
1. Reduced local impact on ecosystems. Maintain and develop ecosystems	• Eutrophication • Biodiversity • Eco-toxicological impact • Land use	Crop rotations, plant protection, management and utilization of bio-diverse semi-natural pastures, manure management, plant nutrient supply, local field management, feed rations
2. Optimize plant nutrient use and supply	• Eutrophication • Acidification • Use of minerals (phosphorous) • Land use	Manure management, plant nutrient supply, crop yields, feed rations
3. Reduced climate impact	• Climate change • Use of fossil fuels • Land use (less land use give space for e.g. bioenergy production).	Energy use, plant nutrient supply (e.g. use of recirculated nutrients), transports, manure management, feed rations, crop yields

3.2. Solution scenarios

Using the goal scenarios above, solutions for each part of the supply chain were developed, so called solution scenarios. For pig production, the most important aspects are feed efficiency, feed composition, production efficiency and manure management. For the post-farm chain the possible improvements are smaller, but include less wastage in industry and distribution, reduced energy use and shifting to bioenergy. Also optimized transports can be of interest. In the project all details are described and quantified, but due to space limitations we only present the most prominent changes for each scenario. Below, the solution scenarios for the pork supply chain are presented, using the reference scenario as a basis. We should keep in mind that solutions should be possible to implement within a ten year time frame, which limits the possibilities to include more drastic changes.

3.2.1. Solution scenarios for the animal production system

Results for the animal production part of the pork chain are presented in Table 3. As can be seen in Table 3 there are many similarities between the three solution scenarios (for other agricultural products there are more differences between scenarios). This is a result of that there are no conflicts between the goal scenarios in this particular part of the production system. The only main difference between scenarios is the feed composition. In all scenarios the basis is grains (wheat, barley, and oats). In the reference scenario the protein feed is mainly soy bean meal and some rape seed meal, in Scenario 1 the protein feed consists of rape seed meal and faba beans. Scenario 2 uses mainly rapeseed meal and some soy bean meal and in Scenario 3 faba beans is the main protein feed, complemented with some rape seed meal. In all three solutions scenarios the use of synthetic amino acid is increased.

A factor that is not changed from the reference is the housing, which is a conventional indoor production with partly slatted floor for fattening pigs and lactating sows, and deep litter bedding for dry sows. Alternative systems as e.g. keeping the animals outdoor might have advantages for some goal scenario, but the short time frame for implementation disqualified such a solution. Also the pig's breed is assumed to be the same in all scenarios.

As explained in the "Methods" section, the description of future scenarios was done by production experts in collaboration with experts on animal welfare, product safety, product quality and economy; hence the solutions scenarios are realistic and based on literature and expert judgments.

Table 3. Solution scenarios for the primary pig production for the reference scenario and the three different goal scenarios developed in the project

	Reference – situation of year 2012	**Scenarios 1,2 and 3**
Surviving piglets per sow and year	23.8	30
Recruitment sows (% replaced annually)	52.5	46
Mortality fattening pigs (30-115 kg live weight), %	1.8	1.2
Carcass meat percentage slaughtered pigs	58.2	60.2
Feed intake sow (incl. recruitment), kg/year	1619	1649
Feed intake, kg per piglet (up to 30 kg/animal)	43	39
Feed intake, kg per fattening pig (30-115 kg live weight)	238	209
Total feed use in the region Västra Götaland (ktons/year)	160	135
Manure management, dry sows	Deep litter	Deep litter
Manure management lactating sows	Slurry, partly slatted floor	Slurry, partly slatted floor
Manure management fattening pigs	Slurry, partly slatted floor	Slurry, partly slatted floor

Since the functional unit in this case was production volumes of pig meat in the region studied, Västra Götaland, for the year 2012, the following changes and consequences were introduced in the animal production in all the solution scenarios;

1. More piglets per sow and lower recruitment of sows lead to lower number of sows needed.
2. Higher meat percentage lead to fewer fattening pigs for the same total meat volume, hence less sows and piglets are needed.
3. Less feed use per fattening pig and more piglets per sow with just a small increase in sow feed leads to a reduced overall feed need.
4. Lower mortality of pig lead to lower number of sows needed.
5. Less nitrogen and phosphorous in manure due to higher feed efficiency (use of synthetic amino acids and phytase for increased phosphorous uptake)

Together, these changes lead to a significantly reduced need for feed, hence arable land, and less plant nutrients in the manure. This gives consequences for the crop production part of the system (below), and is also reduced resource use as such.

3.2.2. Solution scenarios for the feed crop production

The crop production system is connected to all animal production systems in the project (dairy, poultry, beef and pork) as well as to the bread production. Crop production farms produce feed crops that will be used for all animal types, and animal farms can produce a surplus of feed crops to be used at other farms. Our approach was to define type farms for the different production systems; dairy, beef, pork, poultry and crop farming. In the region studied, there are two main types of production areas; plains district with large fields and intensive agriculture, and forest district with smaller and scattered fields and more extensive farming. As a reflection of todays' situation, we assumed the poultry, pork and crop production farming take place in the intensively farmed region, beef production in the more extensive region and dairy in both.

Below, only results for farms producing feed for pigs as well as pigs are presented, a selection made due to space limitations. Actually, some pig feed raw materials come from crop producing farms or is being imported to the region. The main measures in crop production concern crop rotations and use of manure and other plant nutrient supply. Use of pesticides is important for solution scenario 1, as well as farm management options to support biodiversity, such as elements that break up monotonous landscapes, e.g. hedges. Generally, varied crop rotations, with leys and other non-cereal crops, give possibilities to reduce the need for plant protection and facilitate mechanical weeding, as well as reduce the need for mineral fertilizers. Catch crops are introduced as much as possible to reduce nutrient losses and clover/grass ley for green manure. In solution scenarios 2 and 3 the same crop rotation is applied, as well as farm management and plant protection. The reason is that efficient nutrient management and reduced climate impact demand similar solutions. In Table 4 the results for the feed crop production part of the pork chain are presented.

Table 4. Solution scenarios (combination of crop rotation and management measures) for the feed crop production on pig producing farms for the three different goal scenarios developed in the project.

	Reference – situation of year 2012	**Scenario 1 - Reduced local impact on ecosystems**	**Scenario 2 - Optimize plant nutrient use and supply**	**Scenario 3 - Reduced climate impact**
Crop rotation	Winter wheat Spring oats Winter wheat Spring barley	Spring barley [a] Faba bean mech. weed control Spring wheat (undersown) Grass/Clover (green manure) Winter rape, mech weed control Winter wheat [a] Spring oats [a] Spring barley (undersown) Grass ley (grass meal) Spring wheat (undersown) Grass/Clover (green manure) Winter rape, mech. weed control Winter wheat [a] Spring oats [a]	Winter wheat [a] Faba bean [a] Spring barley Winter wheat Spring oats [a] Winter wheat [a] Spring barley Winter oilseed rape Winter wheat Spring oats [a]	Winter wheat [a] Faba bean [a] Spring barley Winter wheat Spring oats [a] Winter wheat [a] Spring barley Winter rape seed Winter wheat Spring oats [a]
Farm management	-	Grass strips without pesticide applications along streams, no pesticide use along field boundaries, “corridors” with permanent grass or hedges across large fields. Cover crops, intensive tillage.	Precision application of fertilizers and pesticides, Cover crops	Precision application of fertilizers and pesticides, Cover crops
Plant protection	-	Reduced use of pesticides, mechanical weed control, use of advanced forecasts for pests	Integrated plant protection, pesticides rather than mechanical weeding	Integrated plant protection, pesticides rather than mechanical weeding

[a] with catch crop

3.2.3. Solution scenarios for manure management

Manure management identified solutions for housing, storage and treatment as well as optimizing manure use in crop rotations. In the reference scenario manure was stored in lagoons covered with a natural crust and spread on grain crops. Scenario 1 had essentially the same handling system as the reference and changed only spreading application rates. Ammonia and greenhouse gas emissions were drastically reduced in Scenario 2 with a housing ventilation filtering system, a tightly covered roof over the storage lagoon, and by acidifying the slurry during spreading. Solutions for scenario 3 centered on anaerobic digestion of manure to produce biogas; which in turn increased the amount of plant available nitrogen and decreased losses to the environment. Furthermore, digestate was acidified to avoid methane and ammonia losses during storage and spreading. For manure application, solution scenarios were restricted to equal crop demands, although scenario 1 and 3 were allowed to apply phosphorus in buffer doses for the next year’s crops. Timing of application was also identified as a feasible measure to reduce losses and was included in the solution scenarios.

3.2.4. Solution scenarios for the post-farm supply chain

This part of the supply chain was divided in four parts; processing, transport and distribution, packaging and by-product management. Some measures needed to be coordinated in order to avoid sub-optimization. One overlapping aspect is the wastage along the supply chain, where all parts contribute, and where actions taken in one part affects wastage at other parts. The issue of wastage is addressed separately. For the post-farm supply chain, few improvement measures involved conflicts between goal scenarios; hence one solution scenario was developed. The only exception was for byproduct management, where different solutions met goal scenario 2 and 3.

For processing it was concluded that major improvements on energy use often requires new investments and possibly new ways of producing. Considering the short time frame applied in the project we focused on incremental changes. The energy savings for the actual slaughter and meat processing was considered to be in the range 20-30% as a food industry average.

For transport and distribution two end-points were included, one large supermarket in a large city and one small grocery shop in a small rural town. The present system is relatively well optimized; hence there were limited improvements for the logistics. However, some minor improvements as fuel efficient trucks and fuel efficient driving were identified which lead to fuel savings in the order of 5%. It is also possible to use biodiesel to reduce greenhouse gas emissions.

For the packaging it was necessary to consider both the function of the packaging (to protect the product to maintain quality and also avoid wastage), how space-efficient the packaged product is in the distribution and finally the environmental impact of the packaging material itself (including waste management). In the reference scenario the smoked and sliced ham was packed in a plastic tray and modified atmosphere was used, 70% nitrogen and 30% oxygen). The pack size was 180 g. In the solution scenario a wallet packaging is used instead, using the same modified atmosphere. The gain is reduced volume per kg product while maintaining the shelf life. This facilitates higher transport efficiency and less use of secondary packaging.

Wastage of raw materials as well as finished products constitutes a large share of post-farm environmental impacts. The causes of wastage are often interconnected with processing, distribution, storage and packaging. On average, 3% of products are lost in the Swedish food industry (excluding retail and households), but variations are large, between companies but also within the same company (Lindbom et al., 2013). The flow of information from retailers to producers is not sufficient and the time before products go out of date is short. This, in combination with less flexible production units within industry lead to overproduction and large storage in industry in order to avoid lack of products when the order comes (empty shelves at the supermarket is not accepted, neither by retailers, nor consumers). Moreover, the food supply chain is a “push-system” in a short time frame, meaning that the primary production cannot adjust its production volumes on short notice; the pigs are growing regardless if there is a demand for pork products the coming week, which adds on to the complexity. Based on a detailed analysis of possible measures to reduce wastage in the food industry (Lindbom et al., 2013) we assume that the wastage can be reduced by 50%, from the present level of 3%.

Food processing generates large volumes of by-products, especially meat processing. Out of a live pig, only around 35% of the live weight of the slaughtered pig ends up as meat and processed meat products (the “Carcass meat percentage” mentioned in table 3 includes bones etc., hence the difference). The possibilities for using these large flows are restricted by the European Directive of Animal Byproducts (European Commission, 2002) laying down health rules concerning animal by-products not intended for human consumption), focusing entirely on safety issues. In the directive animal by-products are classified in three groups: high risk, medium risk and low risk. High risk waste accounts for around 2% of total by-products from pig slaughter, medium risk waste for 13% and low risk waste the remaining 85%. The high risk by-product is presently treated by incineration, and no changes are possible. The medium risk by-product is presently mainly treated as high risk by-products, but a small share is anaerobically digested without recirculation of residues to farmland. Due to regulations and certification systems in Sweden this will not be changed in the solution scenarios. For the low risk by-product, solution scenario 2 and 3 assumes anaerobic digestion and use of residual slurry as fertilizer on farmland.

3.3. Analyses of consequences of solution scenarios

Since the project works with hypothetical systems it was not possible to use experimentally based methods, so the analyses of consequences were theoretical studies based on available knowledge and also mainly of qualitative nature. The consequences included in this part of the project were; product safety (microbiological), animal welfare and product sensory qualities. Throughout the project the design of the solution scenarios were scrutinized by consequence analyses experts, who gave advice how to avoid problems. Finally a short report for each area of consequence was compiled.

For product sensory quality, the impact of a number of aspects was assessed; feed, breed and possible castration of male pigs, stress during live animal transports, reception and waiting at the slaughterhouse and killing were assessed. For the pig meat, impacts of processing, recipes, packaging and distribution were included. Findings included that a change to medical castration instead of surgical will lead to less risk of boar taint and lower fat content, which is considered positive for the sensory quality.

The microbiological food safety was approached using HACCP (Hazard Analysis Critical Control Point) for the identification of critical points and corrective measures, and predictive modeling for calculation of bacterial concentrations along processing and storage. Six processing lines for pork and smoked, sliced ham were evaluated: ordinary slaughter and deboning of pork, automated slaughter, cold deboning, hot deboning, and processing and storage of smoked ham. Some critical changes were found between the reference scenario and the solution scenarios. The identified effects on microbial safety and quality can be controlled by applying relevant control measures. The optimization of heat processing must be related to the microbial quality of the raw material and the extent and likelihood of post processing contamination.

As animal housing did not change much between scenarios, the animal health and welfare was not judged to be much affected in the solution scenarios compared to the reference. However, one thing was that less straw was used for dry sows in all solution scenarios, which might have a negative impact on animal welfare. This can be managed by better knowledge among animal keepers, if straw is used more strategically, as it has been shown in studies.

The cost for primary production is obviously an important consequence of changed production systems. The cost aspect has been included in discussions on choices of farm sizes, animal husbandry systems (e.g. no outdoor production of pigs), possible feeds to use and crop rotations (these need to deliver decent yields over the cycle). In the next phase of the project (Q3-4 2014), costs for primary production will be quantified along with LCA calculations.

4. Discussion

The overall approach in the project, integrating expert competence from various fields of food and agricultural sciences, was extremely beneficial in terms of developing the solution scenarios. By combining expertise on production systems with LCA competence, detailed production systems that maintained the systems perspective of LCA could be designed. The involvement of experts on consequences on products and systems added very valuable insights in how the design affected e.g. food safety and animal welfare, and what measures could be suggested without jeopardizing these critical constraints. The broad competences also enhance our credibility among stakeholders, since the project group as a whole really understands the limitations of and possibilities for food production in a broad sense. Working in such an integrated and synthesizing manner is a novel approach, and it brought about needs for new ways of working and interacting between researchers in different fields. Efficient communication and organization of the project proved to be most important, together with mindset of individuals in the project group.

Not many critical consequences were identified, partly due to the short time horizon that was applied in the project, but also to the close interaction between researchers that eliminated questionable solutions at an early stage. If the time horizon had been longer, hence open up for more radical changes, the role of consequence assessments would have been prominent.

The ambition was to avoid value-based choices in the project. This was partly accomplished by the structured method, including the definition of goal scenarios and the iterative working process. However, it is impossible to be fully objective - some choices need to be made. One example is the presumption that improved production efficiency is favorable for all goal scenarios, which was based on experience from a number of previous LCA studies and also on the fact that globally arable land is a limited resource and that increased productivity is crucial for global food security. However, this presumption could still be contested. Our assertion on full transparency on all choices made throughout the project will not resolve this issue, but will facilitate fruitful discussions about the results.

The trade-off between “reasonable” solutions and optimized ones is also an issue. On the one hand, stakeholder-relevance in the short term calls for known solutions, but on the other hand, substantial improvements often call for more radical changes. This balance is provided by open-minded thinking on radical changes while maintaining a sound realism.

Finally, a broad approach generally implies new challenges to researchers and to project management. With a large number of highly qualified experts, each with limited overview over the entire project, project goals in each sub-project were vital. For the success of the project it was critical to maintain focus and interest among all project members. Therefore, during the more intensive parts of the project, frequent but short meetings between each sub-group and the project management were scheduled in order to keep up the pace and make the information flow efficiently and, not the least important, in both directions.

5. Conclusion

- The working approach applied in the project was successful in integrating LCA research with food system production expertise in order to deliver results that are relevant to decision makers in the supply chain.
- The consequence assessments brought large value to the project, guiding the design of solution scenarios efficiently and preventing solutions that are unrealistic and counterproductive. Examples are a low yield milk system that was abandoned due to poor competitiveness, and reduced cooling at slaughter, that was left out for product safety reasons. The consequence assessment also gave the results higher credibility among stakeholders in the supply chain.
- The issue of value-based choices is important and needs to be managed and communicated in a transparent way.
- Working multi-disciplinary in such an integrated way puts large demands on project management and communication skills.

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Decision support system model for total cost and environmental impact: a case study of rice packaging in Thailand

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ABSTRACT

This study aims to incorporate a life cycle assessment (LCA) technique with multi-objective linear programming (MOLP) into practice through a case study of a rice packaging system in Thailand. The flexible packaging system chosen for the study is a CPP/LLDPE (cast polypropylene/linear low density polyethylene) bag of rice. The incorporated model is intended to determine an optimal usage of raw materials for rice package manufacturing. The decisions were as the consequence of minimization of two trade-off objectives: total cost and global warming potential. The results gained from this study show that the developed model could be used as a decision-support tool to yield a solution that is a compromise between cost and environmental considerations based on the manufacturer's preference and constraints. The optimal value of total cost and global warming potential are $US 6012.23 (equivalent to 192391.34 THB; $US 1 = 32 THB) and 2791 kg CO_2 eq, respectively. To this end, the slight improvement of lowering total cost and decreasing global warming potential at 0.1% and 1.5%, respectively, is below their typical values incurred in actual production.

Keywords: flexible packaging, total cost, global warming potential, life cycle assessment, multi-objective linear programming

1. Introduction

The basic functions of food packaging are not only to contain the food and provide consumers with information on ingredient and nutrition, but also to protect and preserve food products from outside influences and damages. Presently, there are many innovative packaging materials that are used for food products. However, plastic has gained more attention as the most common packaging material. This is because plastics can have various forms and applications, such as films and sheets, bags, pouches, and bottles. Plastics in flexible form are inexpensive and light-weight with a wide range of characteristics, including flexibility, heat sealability, and ease of print, and can be integrated into production processes where the package is formed, filled and sealed in the same production line (Marsh and Bugusu 2007). More than 90% of flexible packaging is made of plastics, compared to only 17% of rigid packaging (Raheem 2012). In Thailand, demand on the flexible packaging industry is expected to grow in response to the continuous expansion of the food industry, as flexible packaging is increasingly used to replace rigid packaging (Plastics Institute of Thailand 2012).

The environmental impact of packaging has become an important issue of society. This is due to a drastic accumulation of packaging waste becoming a major disturbance for life today. Directives and regulations for packaging wastes have been intensively enacted in an attempt to alleviate the environmental problems and thus inevitably being barred in many countries. As a result, manufacturers have gone on a quest for a good packaging solution for their products. One plausible action contrived on a technological development and innovation movement purposely depleted resource consumption but conversely elevated pollutants in the environment. Also, subduing environmental problems with this innovation may subsequently cause manufacturers relevant costs to increase and that can hurt business competition and profit achievement as a whole. This conflict of interest lead to manufacturer difficulties in decision making where simultaneous goals of economic and environmental aspects are being taken into account and the best alternative is not yet clearly determined (Pieragostini et al. 2011).

Life cycle assessment (LCA) is a technique used for identifying and quantifying the environmental performance of a product or process throughout its life cycle (cradle-to-grave) (Azapagic and Clift 1999). Several studies have employed LCA as a tool for determining appropriate packaging used in different applications. These include a comparative study on environmental impact generated by retort sterilization for glass jars, over the aseptic process for plastic containers (Humbert et al. 2009), and an environmental analysis of the entire life cycle of coffee and butter packed in flexible packaging systems (Büsser et al. 2009). Others are an evaluation of environmental impact of packaging systems for canned tuna meat (Poovarodom et al. 2011); a comparative study on environmental impact of strawberry packaging systems: bio-based polylactic acid (PLA) and petroleum-based polyethylene terephthalate (PET) and polystyrene (PS) clamshell containers (Madival et al. 2009). In addition, the single use thermoform boxes made from PS, PLA and PLA/starch was also studied (Suwanmanee et al. 2012).

Normally, the relationship between inventory data and impact category indicators of LCA methodology is linearly expressed by characterization factors, as follows:

$$E_j = \sum_i Q_{ij} m_i \qquad \text{Eq. 1}$$

where E_j is the indicator for impact category j; m_i is the quantity of emission pollutants i; Q_{ij} is the characterization factor that links emission pollutants i to impact category j (Pennington et al. 2004). However, this technique is based on unconstrained human economic activities, such as market demand, material availability and production capacities (Azapagic and Clift 1998). Therefore, the lack of information on economic effect is a major shortcoming of LCA.

As mentioned, decision-making incorporating environmental consciousness with economic cost for packaging solutions should be considered. An appropriate solution, comprising both environmental and economic aspects, in a packaging system can lead to manufacturer satisfaction. The objective of this study is to present the developed model incorporating life cycle assessment (LCA) technique and multi-objective linear programming (MOLP) for a rice packaging system in Thailand as a case study, aimed at combining two objectives: low total cost and decreased global warming potential.

Linear programming (LP) is based on a linear relationship subject to constraint models. This technique is usually used for solving economic problems and is formulated as follows:

$$Min\ Z = \sum_{j=1}^{J} c_j x_j \qquad \text{Eq. 2}$$

$$s.t. \sum_{j=1}^{J} a_{ij} x_j = or \geq or \leq b_i \qquad \mathrm{i = 1, 2, \ldots, I} \qquad \text{Eq. 3}$$

$$and\ x_j \geq 0 \qquad \mathrm{j = 1, 2, \ldots, J} \qquad \text{Eq. 4}$$

where Eq. 2 represents an objective function and Eq. 3-4 are linear equality or inequality constraints in the system. The objective function Z is a value of overall measure of performance (e.g. total cost); variable x_j is a level of activity j; coefficients c_j are the constant values of performance resulting from activity j; coefficient a_{ij} is the amount of resource i consumed by each unit of activity j; coefficient b_i is the right hand side or limitation of resource i that is available for allocation to activities (Azapagic and Clift 1998; Hillier and Lieberman 2001).

Multi-objective linear programming (MOLP) is the LP problem involving multiple objective functions. Thus, MOLP allows the problem where more than one optimization criteria needs to be satisfied (Ragsdale 2007).

2. Methods

2.1. LCA model

The goal of this study is to develop a multi-objective linear programming (MOLP) model by incorporating a life cycle assessment (LCA) technique. Optimality the use of raw materials for rice package manufacturing is determined as a decision considering variables that are the consequence of a trade-off between two objectives. Minimization of total cost for production and minimization of global warming potential are considered two such objectives.

The model system boundary (as shown in Figure 1) involves the acquisition of raw materials, transportation of raw materials and manufacture of the packaging (cradle-to-gate). In addition, packaging materials, including paper core, oriented polypropylene (OPP) tape and polyethylene (PE) film wrap that are required for packing processes are included in the study.

The flexible packaging system chosen for this study is a multi-layer of cast polypropylene/linear low density polyethylene (CPP/LLDPE) bag of rice. A 5 kg package size is used due to it being the most common size of rice products in Thailand. Finished packaging is in the form of bags on a roll (as shown in Figure 2) where the package can be formed, filled and sealed in the same automatic machine. Thus the functional unit is specified as 12,000 m of bags on a roll (a minimum quantity of production).

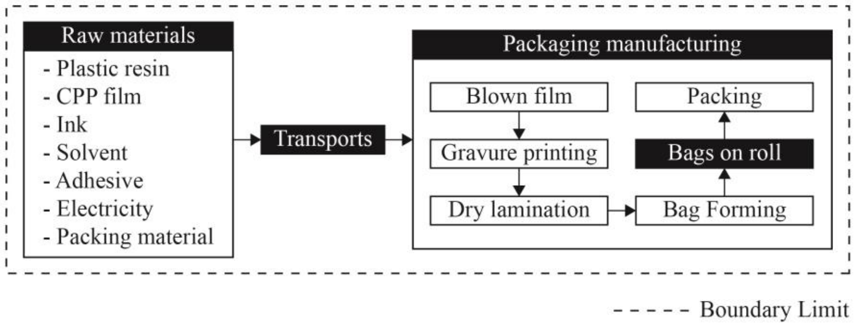

Figure 1. System boundary of the MOLP model.

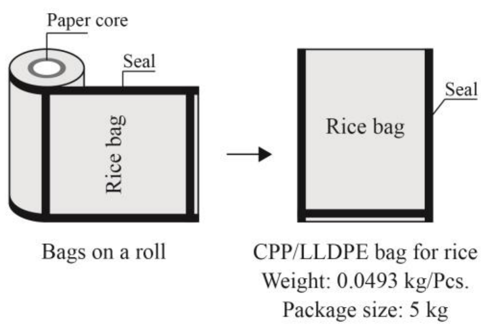

Figure 2. Feature of bags on a roll.

LCA is the most widely acceptable method to assess the environmental impact of a product or process throughout its life span, from cradle-to-gate or cradle-to-grave (Suwanmanee et al. 2012). This study is carried out using life cycle methodology in compliance with ISO 14040 (2006): Environmental management - life cycle assessment - principles and framework.

Life cycle inventory (LCI) data of CPP/LLDPE rice bags was collected from packaging manufacturers. Primary data of input materials, energy consumption, output products, as well as waste generated were collected for each main activity in the production line. The production process includes film blowing, printing, laminating, bag forming and packing. The transportation distance, including vehicle type, for the raw materials is provided (as shown in Table 1). In this study, an assumption on one-way transportation with 100% loading is made for the delivery of raw materials to the packaging manufacturer.

Table 1. Transportation data of raw materials for packaging production

Raw material	Vehicle type	Distance (km)
Plastic resin type A-D	Truck (6 wheels)	130
Plastic resin type E	Truck (6 wheels)	40
Printing cylinder	Truck (4 wheels)	100
CPP film	Truck (6 wheels)	110
CMYK ink	Truck (6 wheels)	100
Other ink	Truck (4 wheels)	50
Solvent	Truck (6 wheels)	50
Adhesive	Truck (4 wheels)	80
Paper core	Truck (4 wheels)	20
OPP tape	Truck (4 wheels)	50
PE film wrap	Truck (4 wheels)	20

Impact assessment is an important phase in LCA framework (ISO 14040 2006) for evaluating the significance of potential environmental impact based on the results of LCI. In this study, impact category of global warming potential (GWP) is chosen as it is relevant to the environmentally important emissions from the production of flexible packaging. The life cycle impact assessment was performed using the CML 2 Baseline 2000 method. Secondary data for emission factor was obtained from the supplier, Thai national LCI database (Thailand Greenhouse Gas Management Organization 2013) and peer-reviewed literature - namely, emission factor of printing cylinder (Ponnak 2011).

In this study, manufacturing cost is classified into four categories: direct raw material cost, direct labor cost, overhead cost and transportation cost. Primary data of direct raw materials cost, machine capability and overhead cost - for example, indirect raw material, indirect labor, depreciation of machine and building, electricity, and others, are approximated and provided by the packaging manufacturer. Direct labor cost is based on the secondary data of minimum wage (Ministry of Labor 2013) and the primary data of number of laborers in each activity of the production line and was obtained from the packaging manufacturer. In addition, secondary data of transportation cost was obtained from the database of cost for freight truck (Department of Land Transport 2009).

A number of constraints can be defined according to LCI as well as specification data of each activity in the production line. First, mass balance is carried out to define the mass balance constraints. This step is normally done in the LCI phase. Second, raw material usage for the CPP/LLDPE rice bags in each activity is limited in compliance with the production capability and performance. Therefore, these constraints can be defined as either equality or inequality. Lastly, the output products in each activity are limited by the functional unit of this study.

2.2. MOLP model

The MOLP model is incorporated with the life cycle assessment (LCA) technique to enable a workable combination of two objectives: minimized total cost for production and minimized global warming potential. The main steps to develop the MOLP model are (Ragsdale 2007)

- Implementing the MOLP model;
- Determining target value for the objectives;
- Determining weighted percentage deviation for the objectives;
- Defining the MINIMAX objective;
- Implementing the revised model and solving.

From the collected data, as discussed in section 2.1, the mathematical model of this study can be formulated as the following.

The raw material usage in each activity of the production line is defined as a decision variable, where the variable x_{ij} represents the quantity of raw material type i in activity j (kg or rolls); i = 1, 2, …, I and j = 1, 2, …, J (as shown in Table 2). In addition, the maximum of weight percentage deviation from target values (Q) is defined as the additional decision variable.

Table 2. List of raw material and activity.

j	Activity	x_{ij}	Raw material
1	Film blowing	x_{11}	Plastic resin type A
		x_{21}	Plastic resin type B
		x_{31}	Plastic resin type C
		x_{41}	Plastic resin type D
		x_{51}	Plastic resin type E
2	Printing	x_{12}	Printing cylinder
		x_{22}	CPP film
		x_{32}	CMYK ink
		x_{42}	Other ink
		x_{52}	Solvent type A
		x_{62}	Solvent type B
		x_{72}	Solvent type C
3	Laminating	x_{13}	Adhesive component A
		x_{23}	Adhesive component B
		x_{33}	Solvent type C
4	Bag forming	x_{14}	Paper core
		x_{24}	OPP tape
5	Packing	x_{15}	PE film wrap

Two objective functions, including minimized total cost and minimized global warming potential can be formulated as follows:

$$Min\ Z_1 = \sum_{j=1}^{J} c_{ij} x_{ij} \quad \text{Eq. 5}$$

$$Min\ Z_2 = \sum_{j=1}^{J} e_{ij} x_{ij} \quad \text{Eq. 6}$$

where Z_1 is the optimal value of minimized total cost ($US); c_{ij} is the unit cost of raw material ($US/kg), labor ($US/hr) or overhead ($US/hr) resulting from raw material type i in activity j; Z_2 is the optimal value of minimized global warming potential (kg CO_2 eq); e_{ij} is the emission factor of raw material (kg CO_2 eq/kg), electricity (kg CO_2 eq/kWh) or transportation (kg CO_2 eq/ton.km) resulting from raw material type i in activity j. However, it is important to note that these objective functions are subsequently are transformed according to the additional constraints of weighted percentage deviation from target values. Thus the MINIMAX objective function will be defined as a new objective function of MOLP model:

$$Min\ Q \quad \text{Eq. 7}$$

$$w_1 \left(\frac{Z_1 - t_1}{t_1} \right) \leq Q \quad \text{Eq. 8}$$

$$w_2 \left(\frac{Z_2 - t_2}{t_2} \right) \leq Q \quad \text{Eq. 9}$$

where Eq. 7 represents an objective function and Eq. 8-9 are additional constraints of weighted percentage deviation from target values. Q is the maximum of weight percentage deviation from target values; the target value t_1 is the optimal value of minimized total cost obtained in single-objective optimization ($US); w_1 is importance weight of minimized total cost; the target value t_2 is the optimal value of minimized global warming potential that is obtained in a single-objective optimization (kg CO_2 eq); and w_2 is importance weight of minimized global warming potential. The weight factors are based on the manufacturer satisfaction. First, weighting factors equal to one are used to yield the same importance of minimized total cost and minimized global warming potential. Later, different relative relevance is considered by adjusting these factors (Pieragostini et al. 2011). In this case, the weight of importance of minimized total cost and minimized global warming potential are defined as 4.6 and 2.4, respectively.

The constraints maintain the mass balance in each activity of the production line. Thus, the sum of all input raw materials must be equal to the total output products, as follows:

$$\left(\sum_{j=1}^{J} x_{ij} \right) - b_j = 0 \quad \text{Eq. 10}$$

where b_j is the output products from activity j (kg).

This constraint maintains the performance of output products in each activity of the production line. The limitations of raw material usage are defined as equality and inequality constraints, as follows:

$$x_{ij} \geq a_{ij} \left(\sum_{j-1}^{J} x_{ij} \right) \quad \text{Eq. 11}$$

$$x_{ij} = d_{ij} \quad \text{Eq. 12}$$

An inequality constraint of raw materials that can be optimized by the MOLP model is expressed in Eq. 11. An equality constraint of printing cylinder, CPP film and packing materials that cannot be optimized by the MOLP model are expressed in Eq.12. In addition, a_{ij} is the minimum percentage of raw material type i in activity j (%), and d_{ij} is the constant quantity of raw material type i in activity j per functional unit (kg).

This constraint maintains the relationship of output products and the functional unit of this study. The limitation of output products can be formulated as follows:

$$b_j = f_j \qquad \text{Eq. 13}$$

where f_j is the limitation of output products from activity j (kg).

The mathematical model is formulated on excel spreadsheets with the addition of solver add-ins (Excel's Solver). The optimality of raw material usage as decision variables for the CPP/LLDPE of rice bag manufacturing is determined as the consequence of the trade-off relationship between two objectives: minimized total cost for production and minimized global warming potential.

3. Results

According to the developed MOLP model, economic cost and global warming potential were calculated for each activity in the production of CPP/LLDPE rice bag to identify feasible usage of raw material while simultaneously minimizing total cost and global warming potential. The optimal solution obtained can serve as a basis to affect the improvement of packaging production.

First, the results of objective functions (Eq. 5-6) are obtained and then normalized to the optimal values of each objective. This can be done by a single optimization of one objective function while neglecting the other. Thus the target values of the two objectives - total cost and global warming potential - are found to be $US 6000.39 and 2780 kg CO_2 eq, respectively. However, it should be noticed here that when the total cost decreases to reach the feasibly minimum value of $US 6000.39, the global warming potential is 0.5% above its target value. Similarly, when the global warming potential decreases to reach the feasibly minimum value of 2780 kg CO_2 eq, the total cost is 1.2% above its target value. These two objectives are in direct conflict with each other. That is, reaching lower levels of global warming potential is associated with incurring greater levels of total cost for the production (Figure 3).

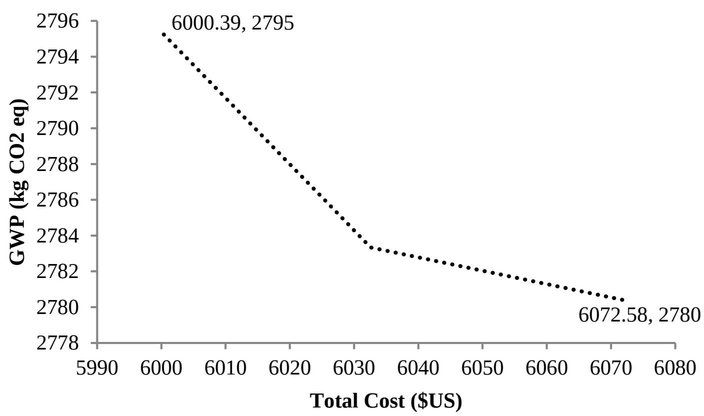

Figure 3. Trade-off relationship between GWP and total cost.

Therefore, the MINIMAX objective is applied to explore the loci on the edge of the feasible region where both objectives can compromise. In the case of the same weight factor (equal to 1) being applied to each objective, to indicate the same level of importance, the result of the MOLP model shows a slight increase in both objectives, approximately 0.3% above the target values. The optimal values of total cost and global warming potential are found to be $US 6018.27 and 2789 kg CO_2 eq, respectively (Table 3). However, it is not usual for industry to consider the cost issue to be of less importance to the decision-making process than the environmental impact. Thus, the weight importance factors of both total cost and global warming potential in this study are adjusted to 4.6 and 2.4, respectively. According to these weights, the optimal value of total cost and global warming potential are shifted to $US 6012.23 and 2791 kg CO_2 eq, respectively. Moreover, the optimal value of total cost shows a lower deviation at 0.2% above its target value. In contrast, the optimal value of global warming potential shows a higher deviation at 0.4% above its target value. This means the manufacturer satisfaction is dependent on how much of one objective they are prepared to give up in order to gain in another.

Although total cost may decrease when more global warming potential is generated, the quantity of raw material usage, as the consequence of the trade-off between these two objectives suggests is the best compromise solution, as is shown in Table 4. Among the five types of plastic resin in film blowing activity, plastic resin type B is found to be increased compared to its typical quantity. This is because it has lower cost and less greenhouse gas (GHG) emissions than the other types of plastic resin. In addition, although the cost of CMYK ink is higher than other solvents used in printing, it is found to increase as compared to its typical quantity because of the limitation in solvents usage. Due to lower GHG emissions there is an increase of adhesive component type B in laminating activity. As a result, the model output suggests that above raw materials should be increased in response to the trade-off relationship between the two objectives. On the other hand, the usage of other raw materials is decreased, subject to their constraints.

Table 3. Optimal values of two objective functions.

Objective function	Weight importance factor	Unit	Optimal value	Maximum of weight percentage deviation from target values (Q)
Minimized total cost	$w_1 = 1.0$	$US	$Z_1 = 6018.27$	0.0030
Minimized GWP	$w_2 = 1.0$	kg CO_2 eq	$Z_2 = 2789$	
Minimized total cost	$w_1 = 4.6$	$US	$Z_1 = 6012.23$	0.0091
Minimized GWP	$w_2 = 2.4$	kg CO_2 eq	$Z_2 = 2791$	

Table 4. Best compromise solution between two objective functions in case of $w_1 = 4.6$ and $w_2 = 2.4$.

Activity	Raw material	Unit	Usage	Compare with typical quantity[a]
Film blowing	Plastic resin type A	kg	198.00	Decrease
	Plastic resin type B	kg	270.00	Increase
	Plastic resin type C	kg	198.00	Decrease
	Plastic resin type D	kg	117.00	Decrease
	Plastic resin type E	kg	117.00	Decrease
Printing	Printing cylinder	Rolls	6.00	Constant
	CPP film	kg	132.00	Constant
	CMYK ink	kg	18.34	Increase
	Other ink	kg	0.47	Decrease
	Solvent type A	kg	1.88	Decrease
	Solvent type B	kg	1.88	Decrease
	Solvent type C	kg	0.94	Decrease
Laminating	Adhesive component A	kg	8.10	Decrease
	Adhesive component B	kg	27.10	Increase
	Solvent type C	kg	45.79	Decrease
Bag forming	Paper core	kg	19.29	Constant
	OPP tape	kg	0.03	Constant
Packing	PE film wrap	kg	1.63	Constant

[a] The typical quantity represents the quantity of raw material which incurred during the real production.

4. Discussion

As the best compromise solution, the optimal values of total cost and global warming potential are found to be $US 6012.23 and 2791 kg CO_2 eq, respectively. Therefore the sensitivity analysis of this MOLP model shows a slight improvement in the optimal value of total cost and global warming potential at 0.1% and 1.5% below their typical values that occur during actual production. However, the system boundary of the model and data availability also implies several limitations. For example, this study focuses on the main activity in the production line of CPP/LLDPE rice bag, but due to a limit of data availability, does not cover the production of CPP film. Likewise some of the data, including labor cost, transportation cost and emission factor is obtained from secondary data. The constraints of raw material usage can be adjusted according to the manufacturer preference.

Thus, there is an opportunity for further improvement in each objective for the minimization of total cost and lowering of global warming potential.

5. Conclusion

Integration of MOLP and LCA can be used to incorporate two aspects concerning economic and environmental issues. The model we developed is applied to determine the optimality of raw material usage for manufacturing the CPP/LLDPE rice bag as a consequence of the trade-off between two objective functions: minimized total cost and minimized global warming potential. According to the results gained from the MOLP model, the best solution shows a slight improvement in the optimal values of total cost and global warming potential at 0.1% and 1.5% respectively, below their typical values incurred in actual production. The compromise solution obtained can identify opportunities to affect production improvement. To this end, the developed model can be used as a decision-support tool to determine a compromise solution to both objectives based on the manufacturer preference and constraints. Also, the model can be used as model prototype for other flexible packaging systems for rice, such as Nylon/LLDPE and PET/LLDPE (polyethylene terephthalate/linear low density polyethylene) in the future.

6. Acknowledgement

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Losses in the supply chain of Swedish lettuce – wasted amounts and their carbon footprint at primary production, whole sale and retail

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ABSTRACT

The waste flow of Swedish iceberg lettuce was followed through the value chain from field to retail shelf. The study also included estimations of the carbon footprint of the waste at the different stages. At the farm level 3 tons of high quality lettuce heads were wasted per ha and year (compared to 19 tons harvested), corresponding to ca 1100 tons CO_2-e per year at a national level. Ca 50% of the lettuce sold in Swedish retail stores is domestically produced and 50% imported. The carbon footprint of the wasted Swedish iceberg lettuce at retail level was estimated to 1500 tons CO_2-e per year at a national level. The conclusion was that the losses at the retail stage were of higher importance than the losses occurring during primary production for the lettuce supply chain, and that measures therefore should be targeted primarily to the retail sector.

Keywords: carbon footprint, food supply chain, food wastage, horticultural production

1. Introduction

1.1. Background

The Swedish National Food Agency has an official assignment to act for reduced food wastage in the Swedish food supply chain. As part of the background material for a new proposed environmental target on reduction of food waste, the Swedish Board of Agriculture initiated a study on the losses, and their carbon footprint (CF), during primary production of lettuce. The current paper highlights the main results of the study (Strid et al., 2014), which also includes the whole sale stage, and merges these results with findings from a study of the retail stage (Eriksson, 2012). The losses of lettuce in retail stores, and their carbon footprint, were studied by the authors in a previous project. By combining these studies, the flow of lettuce can be followed through the value chain from field to retail shelf, and the carbon footprint of the waste can be estimated at the different stages. This type of supply chain study, where the losses are quantified and environmentally evaluated, is likely to be of importance in the policy making for reduced food waste.

1.2. Swedish production and distribution of iceberg lettuce

This subject is described in detail in the background report: Wastage of iceberg lettuce during primary production and whole sale in Sweden (Strid et al., 2014), whereas some highlights are presented here. Iceberg lettuce is mainly produced in southern Sweden, amounting 2012 to 33 400 tons/year. Normally, two or three harvests can be taken each season. Packing can be done either directly at the field - by picking, quality checking, wrapping in plastic bags and packing in cardboard boxes placed on a slowly moving tractor in front of the workers – or at packing tables at the farm center. In the first case, outer leaves and rejected heads are dropped on the field and eventually ploughed down, whereas in the second case the rejected heads are sorted out at the table, and later brought back to the field as compost. The boxes with lettuce are usually continuously moved to a cold storage and then transported away from the farm the same day. At the whole sale distribution center the lettuce is normally loaded on trucks for further transport within a day. The supply chain can schematically be described as in Figure 1.

1.3. Purpose of the study

The present study aims at describing the waste flow of Swedish iceberg lettuce through the value chain from field to retail shelf. The study also aims at estimating the carbon footprint of the waste at the different stages, taking into account that the remaining product flow shrinks as losses occur along the supply chain.

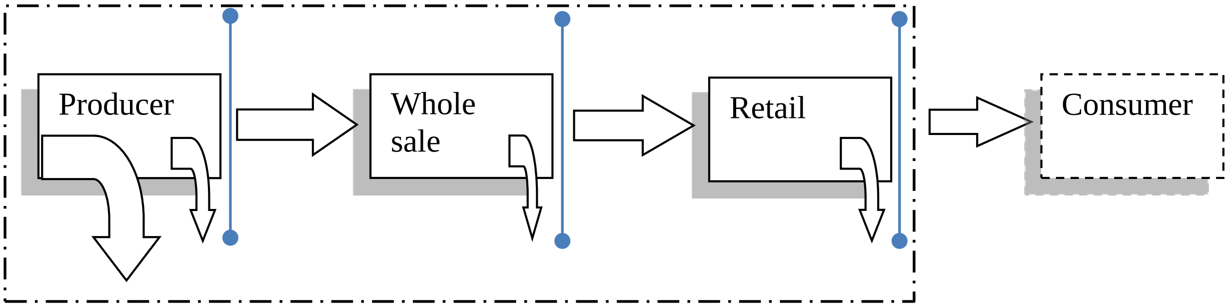


Figure 1. Schematic illustration of the supply chain of Swedish iceberg lettuce, and the system boundaries of the study. The fat waste flow represents un-harvested biomass, and the thinner, losses of prime heads. The vertical lines represent the CF check point belonging to the different stages: Ex-farm, Ex-whole sale and Ex-retail.

1.4. Scope of the study

The study covers primary production of iceberg lettuce, distribution to and waste at whole sale distribution center and distribution to and waste at retail stores, as illustrated in Fig. 1. At the farm, only the waste of high quality heads was considered, and not the type of waste occurring at the field during harvest (e.g., peeled off outer leaves and damaged heads left on the field), since this was defined as production losses instead of food waste. This type of waste can, however, be of interest in other studies, looking at valorization options for the produced biomass. At the whole sale and retail stages, all discarded lettuce was considered. The waste occurring in the households was not addressed in this paper. The study follows the lettuce produced in Sweden as this moves forward in the supply chain. About 50 % of the lettuce sold in Sweden is domestically produced and 50 % imported each year. The normal route for the imported lettuce is via the large distribution centers, and then further to, among others, retail stores. The flow of imported lettuce is not included in this study, but would roughly double the wasted amounts and thereby the CF of the whole sale and retail stages, if it was.

2. Methods

This paper merged and scaled up to national level the results from two studies on iceberg lettuce waste (Strid et al., 2014 and Eriksson, 2012), giving an overview of the losses in the value chain from primary production to retail shelf. The losses were also assessed for their carbon footprint, allowing the different stages of the supply chain to be compared. This was done by applying the product flow's CF at each stage to the wasted amount, thereby answering the research question: "How much extra environmental burden is caused by the losses, assuming that the lost lettuce is replaced by other lettuce produced under the same conditions?"

Each part of the study is described below. The methods used in the two waste inventory studies are briefly described in section 2.1, 2.2 and 2.3; for a full description of the methods, see the original studies.

2.1. Waste during primary production (Strid et al., 2014)

The method used for the study on losses during primary production was a combination of a field study on 5 farms in southern Sweden during the harvest period 2013, and an interview study with the same farmers. The farms practiced direct packing on the field as their harvest method. The biomass left on a number of test squares at the harvested fields was weighted one hour after harvest. During the interview it was explained that 10-20 % of the fields usually are never harvested, mainly due to mismatching orders (if the lettuce gets to old it loses quality and cannot be sold). The losses during primary production was hence accounted as 15 % (mean of 10-20 %) of high quality lettuce ready for harvest, but never harvested. The hectare harvest before dismissing part of the fields was on average 22 tons/ha, thus leading to 18.7 tons actual harvested heads and 3.3 tons not harvested heads. The total biomass left on the fields after harvest was 34 tons/ha, of which 3.3 consisted of the un-harvested heads, and the rest of outer leaves, malformed or damaged heads.

2.2. Waste during whole sale (Strid et al., 2014)

For the whole sale stage, some of the main whole sale organizations were interviewed. The waste was of two kinds: some smaller losses during the normal handling at the whole sale storage and some larger as a result of the quality control of the incoming lettuce, i.e., rejections to the producer, but physically a waste at the whole sale stage. The waste level used for the study was 3.0 %, of which 2.7 % was due to rejections and 0.3 % was due to in-storage losses.

2.3. Waste during retail (Eriksson, 2012)

The waste during the retail stage was based on a Swedish research project on retail food wastage (www.slu.se/foodwastage), which has access to a database covering weekly sales and waste of perishable products during 2010-2012, per item, for 6 stores belonging to one of the larger retail chains in Sweden. In this study both in-store and pre-store waste were assessed, representing the products lost at the retail shelf and those lost already when the products are subject to quality control at delivery. The waste of lettuce was recorded as 10.7 % of the incoming flow, of which 7.3 % was pre-store waste and 3.4 % in-store waste, as documented by Eriksson (2012).

2.4. Up scaling to national level

For the primary production stage, the wasted 3.3 tons of ready heads in relation to 18.7 harvested heads per ha was scaled up to national level according to the mean national harvest 2012, which was 33 400 tons. This led to that 5900 tons per year of produced lettuce never left the farms.

For the whole sale stage, it was assumed that all iceberg lettuce produced in Sweden passes the whole sale stage, and that 3 % of this falls out as waste. 3 % of 33 400 tons equals 1000 tons of lost lettuce, and implies that 32 400 continues to the retail stage.

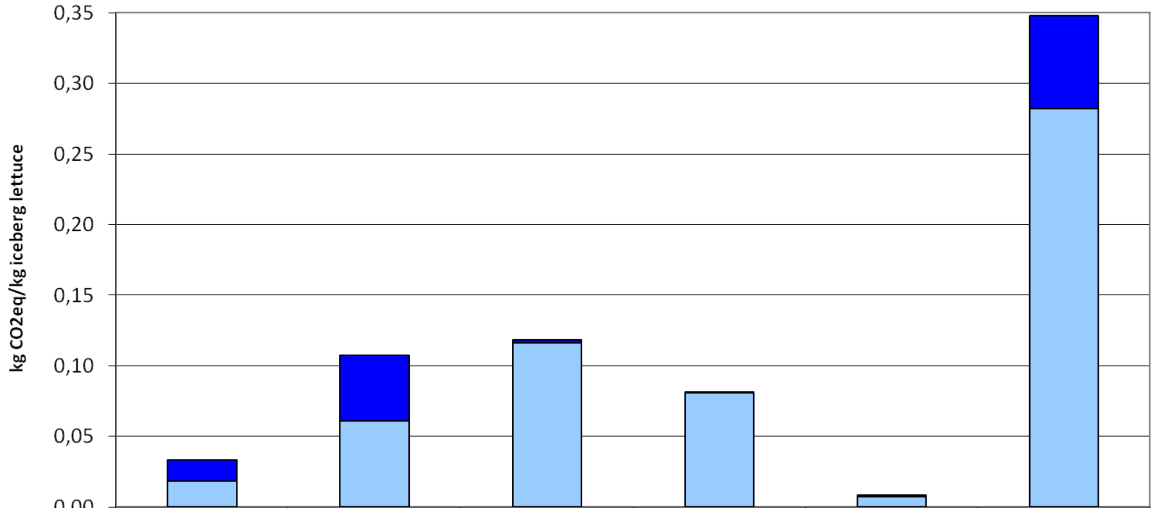
About 50 % of the lettuce sold in Swedish stores is domestically produced and 50 % imported each year. For the retail stage it was assumed that all domestically produced iceberg lettuce passing the whole sale stage also passed the retail stage. This led to a waste of lettuce by 10.7 % of 32 400 tons, i.e., 3500 tons wasted per year for the retail stage, and 28 900 tons passing on to customers.

2.5. Carbon footprint of losses

The last part of the study was to estimate the CF of the losses occurring at each stage. This was done by keeping track of the CF of the outgoing product flow at each stage and multiply this with the wasted amount at the same stage. It was assumed in this study that the wasted lettuce did not have any other offset, e.g., no biogas was produced from the waste. On the other hand, the energy use for transporting the lettuce to waste management facilities at the whole sale and retail stages were not included either. For simplification, these two factors were assumed to even out each other.

2.5.1 Input data used for Carbon footprint calculations

For the carbon footprint data on lettuce production, a Swedish study on emissions of greenhouse gases from production of 17 horticultural products up to and including retail stores were used (Davis et al., 2011). In this, the greenhouse gas emissions at the different stages of lettuce production and distribution were assessed (see Figure 2).



	Fertiliser production	Cultivation	Packaging	Transport	Retail	Total
N2O	0,02	0,05	0,00	0,00	0,00	0,07
CO2	0,02	0,06	0,12	0,08	0,01	0,28

Figure 2. Carbon footprint data of Swedish iceberg lettuce used as input data in the present study. Source: Davis et al., 2011.

However, in this study the waste levels were lower than those found in the more explicit waste studies by Strid et al. (2014) and Eriksson (2012). The waste levels during primary production in Davis et al.'s study were based on questionnaires, where the growers were asked questions about harvest residues and storage losses per hectare. The losses were stated to be 1 ton per ha, in relation to the harvest of 23 tons per ha, and the loss was assumed to be composted and returned to the field. This gives a waste ratio of 4.2 %. There was no question about the share of un-harvested fields, as in the study by Strid et al. (2014), which may have underestimated the actual losses. The waste level during whole sale was not assessed (thus assumed 0 %), but the waste level at the retail stores was included, and derived from Gustafsson, 2010. In Gustafsson's study, the waste at retail stores was obtained from interviews with the store owners, and was assessed to 1.9 % for iceberg lettuce. It is likely that the store managers only regarded the amount that is sorted out and discarded of the in-store products, and not also included the incoming lettuce that was rejected. In a study of Eriksson et al., 2012, it was however concluded that for assessing retail wastage accurate, it is necessary to include both pre-store and in-store wastage. Due to higher confidence in the specific waste studies, new CF values were calculated to reflect these higher loss ratios. The new CF values and the assumptions behind these are described below in section 3.

3. Results

3.1. Adjusting the Carbon footprint values to the assumed product flow

For the cultivation stage, the harvest was assumed to be 19.6 tons instead of 23 tons/ha, taking away the 15 % harvest loss not accounted for. This altered the CF Ex cultivation from 0.15 kg CO_2-e/kg lettuce to 0.18 kg CO_2-e/kg lettuce. Lettuce lost after cultivation, but before packaging, was ascribed this value. For the next stage (Whole sale), the burden from plastic packaging was added, 0.12 kg CO_2-e per kg lettuce, and ¼ of the transport burden, 0.02 kg CO_2-e. This sum was then ascribed to the 97 % of the lettuce making it through the whole sale stage, giving a new Ex whole sale CF of 0.33 kg CO_2-e. For the last stage (Retail), the remaining ¾ of transport, 0.06 kg CO_2-e, was added to the value coming out of Whole sale, and then ascribed to the 89.7 % of lettuce leaving the retail stage for the consumer stage, thus giving a new Ex retail CF of 0.43 kg CO_2-e. The resulting new CF for each stage is listed in Table 1.

Table 1. Calculated carbon footprints per kg lettuce for the different stages of the supply chain

Stage of supply chain	Carbon footprint [kg CO_2-e per kg lettuce] according to Davis et al. 2011	Product flow according to Davis et al. 2011	Product flow in the present study	Carbon footprint [kg CO_2-e per kg lettuce] adjusted to the product flow in the present study
Ex cultivation	0.15	23 tons/ha leaving farm	19.6 tons/ha leaving farm	0.18
Ex whole sale	0.29	23 tons/ha leaving distribution center	19.0 tons/ha leaving distribution center	0.33
Ex retail	0.35	22.6 ton/ha leaving retail	16.9 tons/ha leaving retail	0.43

3.2. Lost amounts and CF of the losses at different stages of the lettuce supply chain

Table 2 gives an overview of the study, describing the relative and absolute waste levels and CF at the different stages of the supply chain. In the last column, the estimated CF of the losses at each stage is shown. The largest losses on a mass-basis occurred during primary production, whereas the largest carbon footprint was associated with the losses at the retail stage.

Table 2. Lost amounts and CF of the losses at the different stages of the domestic supply chain

Stage of supply chain	Loss percentage	CF of lettuce at different stages [kg CO_2-e per kg product]	Lost amount of lettuce at national level [ton per yr]	CF of losses at a national level [ton CO_2-e per year]
Primary production	15 %	0.18	5900	1060
Whole sale	3 %	0.33	1000	330
Retail	11 %	0.43	3500	1500
Total CF of losses				2900

4. Discussion

Contrary to many other agricultural products (tomatoes, broccoli, milk, meat, etc), but similar to other open field crops, the emissions at the production stage of iceberg lettuce is relatively small, leading to that the packaging and transport instead is relatively large; although the absolute value of the product is small. This explains why the smaller amount wasted at the retail stage still can be more important for impact on climate change than the larger amounts wasted at the farms.

When comparing the results with a life cycle assessment study on British and Spanish lettuce production (Hospido et al., 2009), data on losses or waste could unfortunately not be found. The harvest levels were between 15-27 tons of lettuce per ha for open field production, thereby comparable with both the yield without losses (23 tons/ha) and the yield after losses (20 tons/ha) of the present study. The global warming potential was in the study of Hospido et al. estimated to 0.33 kg CO_2-e/kg lettuce for the domestic UK open field production up to a regional distribution center. This can be compared to the CF at the whole sale stage in the present study, which also was 0.33 kg CO_2-e/kg lettuce. In the UK study some parts were different from the Swedish study: postharvest cooling was included, but packaging was excluded.

For mitigation of the environmental impact caused by lettuce waste, the retail stage is the most important stage to target of the stages covered in this study, and within the retail stage the pre-store waste causes the largest volumes of waste. As described by Eriksson et al. (2012), the pre-store waste occurs when the produce arrives at the store. Since the products are formally rejected, the retail store does not have to pay for them. Rejecting a possibly not perfect product already when it arrives is economically favorable compared to be forced to discard it later. This creates economic incentives for pushing products to pre-store waste. Economic feedback systems, such as lower purchase prices if the store keeps a low pre-store waste pattern, might stimulate the development of creative routines and solutions for keeping the waste low..

The pattern of pushing the cost of waste back to the supplier was also found at the whole-sale stage, where more waste was created as rejections, than from handling of the products at the storage (see section 2.2). For the grower, there is no stage earlier to push the waste to. At farm level, development of other market channels for the lettuce that could not be sold to the primary buyer might be a possible solution for reduced wastage at farms.

For comparison, lettuce wasted at household level was in a British study (WRAP, 2014) around 38 % of purchased volumes (64 000 ton wasted of 170 000 ton purchased), thus being more important than all the earlier stages together, if these figures also holds for Sweden. The household waste is not only larger in volume, but also in its carbon footprint, since more impact has been accumulated. If the CF of Swedish household lettuce waste is estimated to 0.5 kg CO_2e/kg (the earlier 0.43 kg CO_2e/kg + home transport and fridge storage), and 38 % of 28 900 tons yr^{-1} leaving retail is wasted, a rough estimate would give that the CF of Swedish household lettuce waste corresponds to 5500 ton CO_2e per year.

5. Conclusion

The main outcome from this study was the indication on to what extent lettuce is wasted at the different stages of the Swedish supply chain of iceberg lettuce up to and including retail. The largest losses on a mass-basis occurred during primary production, followed by the retail stage and last the whole sale stage. The global warming potentials associated with these losses were highest for the retail stage, followed by the primary production stage and last the whole sale stage. Since the retail stage was responsible for the largest contribution to global warming, reduction measures should be targeted to this sector. Within the retail stage the pre-store waste caused the largest volumes of waste. Investigating the root causes behind the retail pre-store waste would give valuable information for reducing the impact of lettuce waste in a food chain perspective. Also, the amount and causes of household waste need to be investigated in future studies.

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Methodological developments for LCI of French annual crops in the framework of AGRIBALYSE®

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ABSTRACT

In the framework of the French research program AGRIBALYSE®, Life Cycle Inventories of 12 annual crops have been worked out. Specific developments were required to establish the LCI of these crops. Firstly, as cropping management and environmental conditions vary both in space and time, emissions modelling at a national scale is a complex task and induced methodological challenges. Secondly, environmental assessment of a single crop requires taking into account the whole cropping rotation, in particular to assess nitrate leaching emission and allocate nutrient burdens between crops. Hence, a specific work, based on statistical data, was done to develop a nitrate model and to take into consideration fertilization practices into whole rotations. These developments provide a framework to assess emissions at a national scale in consistency with average agricultural practices and field observations. Availability of data can be a limiting factor and determine choices made for agricultural LCA-making. The nitrate model and the allocation method developed for AGRIBALYSE® are two examples of structured and simplified methods based on and parameterized with national statistical data and provide consistent results.

Keywords: annual crops, allocation, LCI, nitrate, rotation

1. Introduction

AGRIBALYSE is a French research program (2010-2013) launched by ADEME and aimed to produce public Life Cycle Inventories (LCI) of agricultural products. It pursuits two purposes: i) build an LCI database to provide data for the environmental labeling of food products and ii) share the data to enable the agricultural and food industries to assess the production chain and reduce environmental impacts (Colomb et al., 2014; Koch P.; Salou T., 2013). In the framework of this program, Life Cycle Inventories (LCI) of 25 product groups have been produced, whereof 12 are referring to annual crops, as cereals, leguminous, oleaginous and industrial crops,. A common methodological framework has been established for all the plant production inventories to ensure a homogeneous database. But at the same time, the crop diversity (annual / perennial, field / greenhouse, French / tropical crops) required specific choices to model at the best the different productions. LCA of crops induce different challenges:

(1) *Data collection:* for French cereals, leguminous, oleaginous and industrial crops, detailed and representative statistical production related data are available by crop for the main production systems from national administration or technical institutes. However, this is not the case for organic production. Moreover, few data on soil and climate conditions are available in these surveys.
(2) *Allocation of inputs and direct emissions between crops:* emissions depend on practices managed on the whole crop rotation and not only on one crop. For instance, fertilization to one crop may benefit following crops too. Moreover, the emissions of some environmental flows vary in function of the management and features of one crop but also of the preceding or following crops. In particular, nitrate leaching due to the nitrogen fertilization of one crop depends on the quantity applied but also on the intercrop management and on the following crop. Hence, environmental assessment on single crops requires taking into account the whole crop rotation. But statistical data are rarely available on a crop rotation level.
(3) *Calculation of direct emissions*: the estimation of an average value for some emissions, as nitrate, for a given crop at the national scale is difficult as the emissions greatly vary depending on agricultural practices, type of soil and climatic conditions.

The main methodological choices and the database as well as a result analysis for annual crops are described in two other contributions to the LCA Food 2014 (Colomb et al., 2014, Willmann et al. 2014). This article describes the main specific choices and methodological developments that have undertaken to cope the features of

French annual crops and agronomic specificities. The first part discusses the modelling framework. Specific developments for nitrate leaching assessment and nutrients burdens allocations are presented in the two following parts.

2. Modelling framework

For a complete and detailed information on methodological choices, we here refer to the AGRIBALYSE methodological report (Koch and Salou, 2014). For a general description about the modelling framework, system boundaries and the quality control, see Colomb et al. (2014).

2.1. Main methodological framework

LCI for 12 cereals, leguminous, oleaginous and industrial crops have been established. Several production systems for the same product have been described to distinguish different productions in function of their quality or of the mode of production (Table 1). The system boundaries considered in AGRIBALYSE are from cradle to field gate.

Table 1. AGRIBALYSE inventories for cereals, leguminous and oleaginous and main sources for description of production systems

Product groups	Inventories	Main source for description of production systems
Barley	Barley, conventional, malting quality, national average	[1]
	Forage barley, conventional, national average	[1]
Durum wheat	Durum wheat grain, conventional, national average	[1]
Soft wheat	3 Soft wheat grain, conventional LCI: breadmaking quality, 15% moisture / protein improved quality, 15% moisture / national average	[1]
	2 Soft wheat grain, organic (model type), Central Region LCI: after Alfalfa / after fava beans	[2]
Triticale	Triticale grain, conventional, national average	[5]
	Triticale grain, organic (model type), Central region	[2]
Grain maize	Grain maize, conventional, 28% moisture, national average	[1]
Silage maize	Silage maize, conventional, national average	[1]
Rapeseed	Rapeseed, conventional, 9% moisture, national average	[3]
Sunflowers	Sunflower, conventional, 9% moisture, national average	[3]
Faba beans	Faba beans, conventional, national average	[5]
	Spring faba beans, conventional, reduced protection	[5]
	Faba beans, organic (model type), Central Region	[2]
Peas	Winter pea, conventional, 15% moisture	[1]
	Spring pea, conventional, 15% moisture	[1]
Potatoes	4 ware potato, conventional LCI: for industrial use / for fresh market, firm flesh varieties / for fresh market, other varieties / variety mix, national average	[1]
	Starch potato, conventional, national average	[1]
Sugar beet	Sugar beet roots, conventional, national average	[4]

[1] survey on agricultural practices of the French agricultural administration managed in 2006 (2006 SSP farming practices survey), [2] RotAB: project for organic arable crop systems without livestock (http://www.itab.asso.fr), [3] survey of CETIOM, [4] survey of ITB, [5] according to expert opinion

As for most of the production systems, detailed and representative statistical data about agricultural practices are globally available by crop from surveys of national administration or technical institutes, these data were used by preference (Table 1). With the exception of organic production and of some conventional produced crops (faba beans, triticale), those provide detailed information about the different crop operations and applied inputs from the harvest of the preceding crop to the harvest of the surveyed crops and the harvest. If necessary, these data were completed by expert opinions in particular to distinguish agricultural practices in function of their quality and to adjust data to be representative for the period 2005-2009. On the other hand, no data on soil and climate conditions were available in these surveys. In the absence of representative statistical data, study cases from the project RotAB have been used to establish LCI for organic productions. So, the aim of these inventories is not to represent the French organic production but to provide an order of magnitude for one example of organic arable system production without livestock.

Emissions associated with inputs are based on existing data, mainly from the database Ecoinvent® V2.2. Some input references have been adapted to French specificity when possible as for example fuel consumption by tractors during field work, transport of inputs and fertilizer product.

2.2. Identification of models for the calculation of the direct emissions

The substances quantified have been selected following international standard recommendations, knowledge about the contribution of the substance emitted by agricultural activities on environmental impacts and the existence of valid models. Each flow was calculated with a specific model chosen as the most suitable according to the objectives and limits of the program. Such models should not require input data that are too difficult to collect, and should be validated for France and recognized internationally. We also considered whether models simulate the effect of agricultural practices on emissions.

For some flows, different models used for LCA, national inventories, agronomical research or development have been identified in particular for NO_3^-, N_2O or NH_3 emission assessment. Mechanistic models allow the assessment of evolving changes in farmer practices but are often applied to a relatively small geographical region, assuming a homogeneous environment with respect to soil and climate. Their application at a larger scale is difficult as input data to models, including cropping management and environmental conditions, can vary both in space and time. In the meantime, no or few data on soil and climate conditions were available in the surveys on farming practices. Hence, models used to assess N_2O or NH_3 emission for national inventories (IPCC 2006b Tier 1; EMEP/EEA 2006, 2009 Tier 2) were finally chosen. For NO_3^- emission, a specific empirical model has been developed based on a risk analysis approach (see 3. Nitrate leaching modelling).

Only one model meeting AGRIBALYSE requirements was identified for some flows as phosphorus (SALCA-P), heavy metals (SALCA-ETM) or eroded soils (RUSLE). The application of these models for AGRIBALYSE required some adaptations, in particular taking into account French production conditions and using them at a national scale, while they have been established for applications at field scale. Hence, some parameters have been adapted when possible. For instance, average soil and climate parameters at regional scale have been assessed basing on INRA soil and metrological database to implement RUSLE. However, as data were missing, it appeared that the adaptation of SALCA-P to the French context was not possible. Some parameters related to plots, as slope, have been assessed by expert judgment. To assess the relevance of this model on French situation, a specific work was done to compare results to experimental data (Willmann et al., 2014).

Some flows were not assessed as no valid model has been identified. It is the case of particulate emissions other than ammonia, as the data currently available in France and Europe was considered to be insufficient to take satisfactory account of these emissions. Among the various NOx gases only NO was considered for direct flows, owing to the lack of appropriate models for the other gases.

3. Nitrate leaching modelling

3.1. The aim of the model

Different mechanistic field-models have been identified to assess either only NO_3^- emission (eg: the model DEAC, described in Cariolle 2002 and Cohan et al., 2011; or SALCA-NO3, described in Richner et al. 2006), or NO_3^-, NH_3 and N_2O emissions (the model STICS, described in Brisson et al. 1998 and Syst'N, being published). In the meantime, their application at region scale is difficult. Schnebelen et al. (2004) developed and assessed an upscaling approach of STICS to model nitrate leaching at regional scale. This approach has been proven effective at the scale of a small agricultural area (i.e; 526 km2) but is likely to introduce additional errors, as it is based on assumption of homogeneous crop and soil parameters on simulation units. Moreover, the application of this approach at the French national scale is a lengthy and data-intensive process.

Very simple models that estimate nitrate leaching as a fraction of applied fertilizer (IPCC 2007; Miller et al. 2006; Powers 2007) are too crude to compare production systems or assess effects of changes in farmers' practices. Bentrup et al. (2000) estimated nitrate leaching on the basis of estimated post-harvest soil mineral nitrogen content, but some input data for the model (nitrogen mineralization and immobilization in the soil) are difficult to quantify at large scales. Basset Mens et al. (2006) developed a method combining a risk analysis approach, based on the risk analysis proposed by Cattin et al. (2002), with regional leaching data. In this study, a model was specifically designed to assess some production systems in the Brittany region in western France, and the method therefore is not valid outside this region. However, this approach has been proven efficient at providing an average estimation of nitrate leaching for a specific region and it is based on easily available data regarding farmers' practices and soil parameters. This method could therefore be applied at a national scale after specific development.

3.2. General principles of the model

The COMIFER model proposed by Cattin et al. (2002) is a qualitative simplified approach that is applicable at a plot scale to qualify the risk of leaching. It takes account a "crop risk" and an "environment risk" (depending on the quantity of water percolating through the soil and the mineralization conditions).

Concerning the "crop risk", the COMIFER model classifies conditions according to the following criteria, in order of importance.

1. Period without presence of vegetation able to absorb nitrogen (depending on the following crop and the sowing of an intermediate crop)
2. Capacity of the following crop to absorb nitrogen in the fall (depending on the following crop)
3. Application of organic fertilizer in the fall (C:N ratio < 8)
4. Quantity of nitrogen provided by crop residues (depending on the crop studied and the management of residues)

The COMIFER model defines the "environment risk" by the combination of two criteria: i) the soil drainage index and ii) the organic matter content of the mineralizing soil layer.

Then, it classifies the risk according to various "crop risk" x "environment risk" combinations.

3.3. Specific developments

ARVALIS managed different developments based on this model for its implementation in AGRIBALYSE:

1. A weighting system for the different classification criteria was drawn up by expert opinion on the basis of the model to estimate the risks for crop conditions not covered in the COMIFER classification.
2. The COMIFER provides a risk classification according to the crop and environment risks. Each level has been associated with a nitrate leaching amount, based on expert opinion.

Table 2. Nitrate leaching amount (kg $N\text{-}NO_3^-$/ha) according to the crop and environment risks.

		« crop risk »				
		1	**2**	**3**	**4**	**5**
« soil risk »	**1**	5	10	20	25	30
	2	10	15	25	30	40
	3	15	20	30	40	50
	4	20	30	40	55	60
	5	30	40	40	60	80

3. The balance between nitrogen supply and the crop requirements is not a parameter used in the COMIFER model. This was based on the assumption that there was no excessive nitrogen fertilization on the previous crop which could generate an excessive increase in the nitrogen content of residues. On the one hand, many studies (Chaney, 1990; Richards et al., 1996; Machet et al., 1997) have shown that when N inputs are lower than the crop requirement, the N application has a very little effect on the nitrate amount that is available after harvest

and potentially leachable. On the other hand, excessive N inputs can potentially increase the amounts of N leached. These depends on cropping practices, soil type and climate conditions (Lacroix et al., 2006).

First, the effect of the excessive N inputs (N inputs minus crop N requirements) on the residual soil mineral nitrogen content (SMN) after harvest has been assessed from the relationship established by the COMIFER (1997), based on 400 measures. Then, the surplus amount of nitrate leaching was assessed, based on the supplementary SMN and parameters on leaching fraction. As the effect of excessive N-fertilization was merely assessed on the experimental sites, these fractioning parameters have been established for each "environment risk" by expert opinion (Table 3).

Table 3. Fractioning parameters to assess the effect of excessive N-fertilization from the increase of SMN and nitrate leaching results for an excess of 50 kg N/ha

		Soil risk				
		1	**2**	**3**	**4**	**5**
% of the supplementary SMN that is leached		0.5	0.6	0.7	0.8	0.9
Supplementary nitrate leaching (kg N-NO_3^-)	**cereals and oilseed rape**	7	8	9	11	12
	maize and sugarbeet	12	14	17	19	22

The "crop risk" weighting system and assignment of leaching quantities to each "environment risk" x "crop risk" combination were validated by measures on 5 sites for three to fourteen years on contrasting situations. Some simulations have also been made with the DEAC model (Cariolle 2002) in order to obtain references for a very low « soil risk ». The validity of the model at plot scale was also confirmed by a work being published. This work aims to compare three different models applied on two different study cases.

3.4. Implementation of the model for the AGRIBALYSE program

As no statistical data cover both agricultural practices and soil-climate data, different databases have been used (Table 4). It was not possible to localize precisely the agricultural plots from the public survey on agricultural practices but the sampling of this survey was built to provide representativeness at the scale of administrative regions. Hence, nitrate leaching was assessed at the scale of administrative regions, although these regions do not correspond to homogeneous agronomical units.

The crop risk was estimated for the 1,000 plots covered by the SSP for the 2006 farming practices. For each crop and each administrative region, the average risks were estimated for each "crop – following crop" combinations. The average risks for each crop and each administrative region were obtained from these risks for crop combinations weighted by the frequency of these combinations, also estimated with the farming practices survey.

The geographic French soil database managed by the INRA Soil Science Unit, Orleans, was used to estimate the environment risks. It describes a set of soil typology units, characterizing distinct types of soil. The soil typology units are described using attributes defining the nature and properties of the soils (eg: texture, water system, soil parent material, etc.).The environment risk was estimated for each soil typology unit based on soil water retention capacity and climatic data from the Arvalis database covering 84 weather stations over the last 30 years. The areas corresponding to each risk category in each administrative region were then characterized to estimate the average risk.

Average nitrate leaching amount for each crop was estimated at administrative region scale from the average crop risk in the administrative region and the average environment risk of the administrative region. However, for the plots of the farming practices survey, available data were insufficient to estimate a N recommendation and to compare it with the N inputs in order to take into account the effect of excessive inputs. Hence, we managed a sensitivity analysis, considering the frequency of excessive N fertilization from 5 to 20% and N inputs exceeding by 50 kg N/ha the crop requirement.

Table 4. Data sources for the implementation of the model for the AGRIBALYSE program

Input data	Source data
Volume of production of the crop per region	statistic of the French agricultural administration for 2005-2009
Following crop	2006 SSP farming practices survey
Intercrop following the crop studied	
Application of organic fertilizer in the fall	
Residue management	
Excess amount of N inputs	Expert opinion
Soil-climate data: water properties of the soil (characteristic humidity, root depth of crops), meteorological data (rain, potential evapotranspiration), organic matter content	geographic French soil database managed by the INRA Soil Science Unit, Orleans meteorogical: Arvalis database

3.5. Results

We can identify 4 groups of crops.
1) The lowest emissions are observed for sugar beet because of the low soil nitrate nitrogen after its harvest.
2) Sunflower and potato harvested in autumn are mainly followed by soft winter wheat. As the intercrop duration is short, emissions are low.
3) Nitrate leaching amounts are in the same range for pea, oilseed rape, cereals and grain maize. Emissions for barley are lower than those of soft winter wheat as 40% of barley plots are followed by oilseed rape, instead of 14% for soft winter wheat. As oilseed rape has a high capacity of N absorption, this explains this difference.
4) Finally, forage maize emissions are higher because of low residues restitution. As the C/N ratio of maize residues is high, those induce a net organization of soil nitrogen during the following intercrop and then reduce the risk of nitrate leaching.

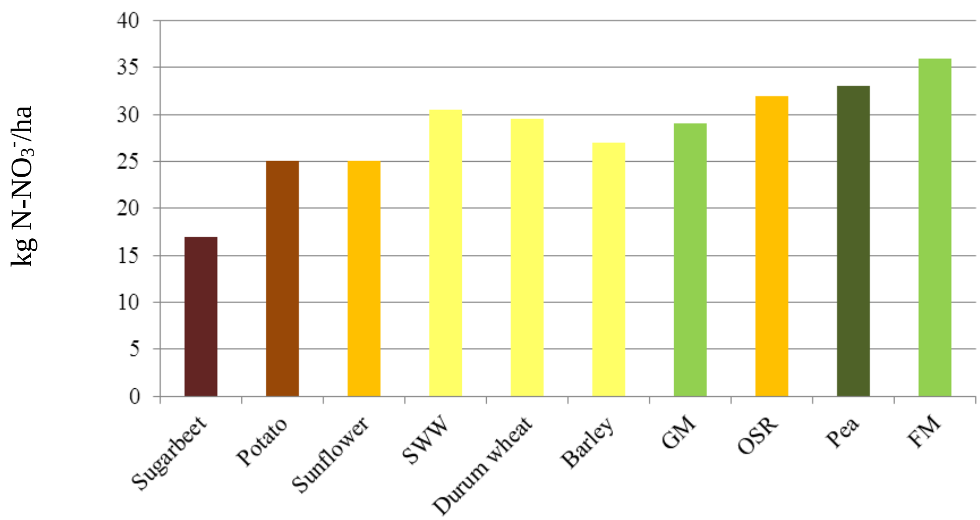

Figure 1. Average nitrate leaching amount for each crop at national scale. SWW: soft winter wheat; GM: grain maize, OSR: oilseed rape, FM: forage maize

As this model only concerns annual crops, its results can be compared to results from the literature at the regional scale for regions with a low proportion of area occupied by meadows and perennial crops. Using these results and the crop frequencies in each region, average nitrate leaching amounts per region were estimated and compared to literature results at the watershed scale (Table 5). The estimation of among 30 kg $N\text{-}NO_3^-$ leaching under root-zone is confirmed by these published results.

Table 5. Comparison of results at regional scale using the AGRIBALYSE methodology scale with bibliographic references

Watershed	Source	Years	Flow kg N/ha/an	Description	AGRIBALYSE results for the concerned regions, kg N/ha/an
Seine	Ducharne et al. 2007	1990-2000	30	NO_3^- leaching amount under root-zone estimated by STICS	26-30
Marne	Ledoux et al. 2007	1995-2000	20-33	NO_3^- leaching amount under root-zone estimated by STICS-MODCOU	34

The validation by experimental results shows that the model is relevant to estimate nitrate leaching at plot scale from easily available data. The application at administrative region hides a wide diversity of situations but provides consistent results for average nitrate leaching amount at regional scale.

4. Allocation of nutrient burden between crops

4.1. Principles and allocations for cropping sequences in the AGRIBALYSE database

It is difficult to allocate the impacts of a production system to each crop in a cropping sequence because certain practices may involve several crops in the rotation system and certain emissions depend on the practices and characteristics associated with a crop as well as on the practices and characteristics associated with previous or following crops.

The ILCD Handbook (JRC and IES 2010) recommends taking account of the nutrients remaining in the system after a crop has been harvested as a co-product of this crop and continues by extending the system or by allocation. Different allocation rules have been identified in bibliography (Audsley et al. 2003; Blonk 2010; Gac et al. 2006; Van Zeijts et al. 1999). However, these references rarely lead to common accepted rules in term of methodology or their suggestions are not always easy to apply because of data availability (Gueudet et al., 2012).

It is the case for allocation of Phosphorus (P) and potassium (K) inputs. For instance, in some studies, they are allocated in function of the nutrients exports (Gac et al. 2006; Van Zeijts et al. 1999) or of the recommended quantities (Audsley et al. 2003; Blonk 2010). We chose to allocate them in function of crop exports as the allocation based on recommended quantities require data that are not easily available at the national scale (PK soil content, PK input during the past few years).

Concerning the allocation of nitrate, Powers (2007) develops a method to allocate the flux between the different crops in function of the fluxes measured on fields cultivated on single crop. However, this is not applicable for every situation as some crops cannot be cultivated as single crop. Audsley et al. (2003) prefer to allocate these fluxes half to preceding crop and half to the following crop. William et al. (2006) allocate the total leaching fluxes at the rotation scale to the different crops in function of the post-harvest N surplus. These two options require data on precise practices data on the whole rotation that were not available at the national scale. Hence, in this study, we chose to allocate the estimated leaching flux to the crop 1.

Table 6. Allocation rules used for cropping sequences

Element	Comments	Allocation rule
Phosphorus (P) and potassium (K) inputs	These are immobile in the soil. Some farmers use residual nutrients by applying P and K fertilizers to one crop only in quantities sufficient to supply the needs of following crops. P and K inputs remaining as crop residues were considered as available for the following crops.	The impacts associated with the production of these inputs and emissions (P, PO4, ETM) related to their application are allocated to each crop proportionally to their nutrients exports, assessed from the harvested yield and PK contents of harvest. Sources: COMIFER farming practices survey and tables of crop exports
Phosphorus (P) loss	Main of phosphorus loss is induced by soil erosion. Soil erosion by crop was assessed by RUSLE, which takes into account occupation time and agricultural practices on each crop.	No allocation was managed on soil erosion and P loss.
Organic nitrogen	Only a fraction is directly available to the crop to which the organic nitrogen is applied. The rest contributes to a stock of organic matter, which could benefit all crops in the rotation.	The nitrogen available for the crop to which the fertilizer is applied (PAN) is allocated to that crop. The rest is allocated to all the crops in the rotation. Sources: Farming practices and the mineralization dynamics of organic fertilizers from the CASDAR project "Sustainable soil management"
Mineral nitrogen	The amounts of nitrogen applied in mineral form are directly available for the crop to which the fertilizer is applied.	The impact of production and the emissions of the nitrogen applied to a crop in mineral form are allocated in full to this crop.
Intercrop-nitrate	Residual nitrates remain in a soil after a crop has been harvested. These may be used by the following crop but a fraction may also be leached.	The impact of nitrate production and emissions between crops are allocated to the previous crop.
Nitrogen from crop residues	Crop residues may constitute a source of nitrogen for the following crop(s). They may also produce N_2O emissions evaluated using the methodology of IPCC Tier 1.	The impact of nitrogen contained in crop residues and the N_2O emissions from these residues are allocated to the crop which produced these residues.

4.2. Allocation of organic N, P and K inputs on the basis of the 2006 SSP farming practices survey

Organic N, P and K inputs were allocated to all crops in a cropping sequence according to the rules outlined in table 7. This type of allocation required a detailed knowledge of the fertilizers applied and the yield for each crop in the rotation system. There was little statistical data for cropping sequences and the information that was available did not cover the production of all the crops studied in AGRIBALYSE. The 2006 SSP farming practices survey covered crops and not cropping sequences. However, it had the advantage of covering most of the main production regions for about ten major crops. It also gave information on the history for each plot and in particular details about previous crops. The year 2006 was considered representative of fertilizer applications during the reference period 2005-2009.

The succession of crops grown in the 14,000 plots studied was known for the period 2001 to 2005. An analysis of this data showed nearly 4,000 different cropping sequences. This diversity and the size of the samples did not make it possible to reconstitute fertilization practices for each type of cropping sequence. To take account of this diversity, these rotation systems were grouped together as "cropping sequence groups" using appropriate statistical optimal matching methods (Gabadinho et al., 2011). The 4,000 rotation systems were grouped into 34 major cropping sequence groups, depending on the dominant crops and production region (Jouy and Wissocq, 2011; see
Table 7 for example). This made it possible to take account of the differences in the application of fertilizers for a given crop depending on the rotation system and region, based on SSP 2006 data. After allocation according to the rules in Table 6 for each crop within the cropping sequence groups, a French average input of organic N, P and K was calculated for each crop.

Table 7. Example of cropping sequences identified in the Northern region

Region	Cropping sequences	Area in 2006 (ha)
Northern	Sugarbeet (18%), winter wheat (50%), potato (8%)	719 000
	Silage maize (24%), temporary grassland (10%), winter wheat (39%), barley (13%)	561 000
	Oilseed rape (23%), winter wheat (44%), barley (23%)	551 000
	Winter wheat (43%), barley (13%), leguminous	254 000
	…	

4.3. Results

The allocation rules modify significantly the nutrient amount attributed to each crop (Table 8). For instance, organic manure is spread by preference on silage maize as this crop uses efficiently organic nitrogen. In the meantime, only a fraction is available for silage maize. Hence, organic N amount allocated to silage maize is much less than the spread amount. High amount of K_2O is applied on potato in particular as this crop requires amount superior to this exportation. A fraction of this amount is allocated to the other crop as this K_2O “excess” remains available for the following crops.

Table 8. Nutrient attributed by crop before (B) and after (A) allocation, example on a few crops

	kg organic N / ha		kg mineral P_2O_5 / ha		kg organic P_2O_5 / ha		kg mineral K_2O / ha	
	B	A	B	A	B	A	B	A
Silage maize	166	114	24	33	73	46	14	4
Soft winter wheat	11	18	25	39	7	19	42	10
Spring pea	-	9	29	22	-	7	46	19
Starch potato	55	41	47	25	21	12	202	161

Their effect was particularly significant on three indicators (Figure 2):

- the global warming (IPCC 2007) in particular on maize (variation of 10%) as N_2O emission is function of the applied organic N amount and NH_3 and NO emissions,
- the freshwater eutrophication potential (ReCiPe 1.05H). Field emissions do not significantly vary with allocations as SALCA-P was only sensitive to P_2O_5 spread as liquid manure and as mineral or solid manure. In the meantime, mineral P_2O_5 production contributes from 30 to 50% of the freshwater eutrophication potential, this explains the significant effect of allocations on this indicator (variation from 10 to 20%),
- the cumulative fossil and nuclear energy demand (CED1.8) of potato (variation of 7%).

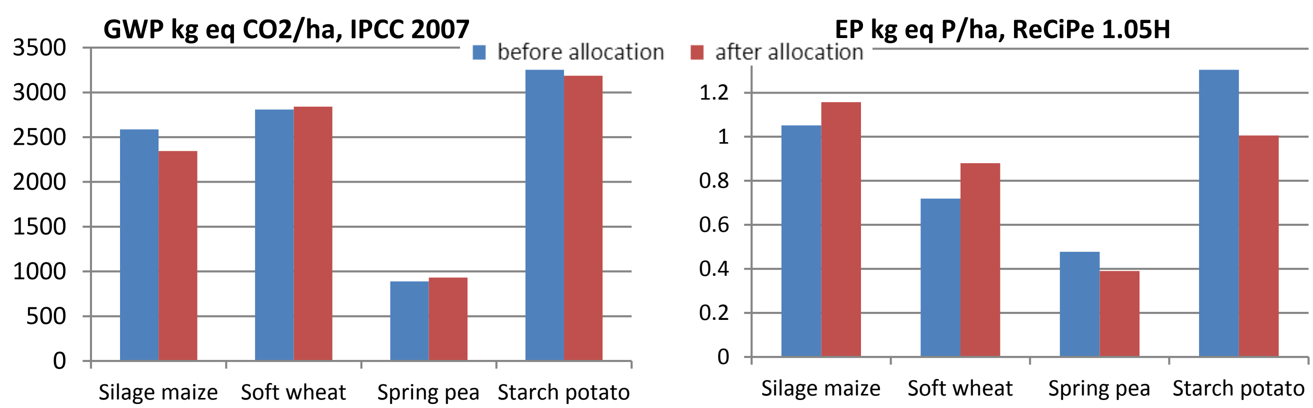

Figure 2: allocations effects on LCA indicators, example of global warming potential (GWP) and eutrophication potential (EP)

5. Conclusions

Substantial work and developments have been carried out within AGRIBALYSE program in order to produce consistent references relative to real agricultural practices and field observations in France. The implemented methods should evolve with improved knowledge.

In France, statistical databases on agricultural practices or on the soil are valuable sources of information for the LCAs of crops. Thanks to their exploitation and crossed with the expertise in the field, AGRIBALYSE was able to establish average references. Further work on typology of production systems would enhance the understanding of the variability of impacts due to the agricultural practices and the natural conditions. However, for some sectors (eg organic production), the statistics are lacking and should therefore be completed. The actual LCIs for these products have a lower representativeness than for other products.

Moreover, the estimation of some flows (eg N_2O emissions, carbon storage in the soil) may be improved soon. Further to AGRIBALYSE program, the project ECOALIM, funded by ADEME and CASDAR, aims to optimize feeding to reduce environmental impacts of livestock. Important work is carried out within this framework to estimate fluxes of nitrogen in water and air and allocate impacts in the crop rotation. But we must not stop there. Other flows (eg phosphorus, pesticides and trace elements) are estimated with a high uncertainty because the available models are not robust enough or due to insufficient data for adapting the models to the French context regarding phosphorus. They must be also subject of works.

Finally, some impacts, such as water and biodiversity, were not covered within AGRIBALYSE, due to a lack of adequate methods or of available data. Both are major issues and will be the subject of research efforts in the future.

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Synergies and trade-offs between the greenhouse gas emissions and biodiversity performances of global livestock production

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ABSTRACT

We provide the first global environmental assessment of livestock production that includes both greenhouse gas (GHG) emissions and biodiversity criteria. We compared performances on these two environmental criteria across scales, commodities (dairy and beef cattle) and production systems (grassland and mixed). To do this, we combined a global model computing the greenhouse gas emissions of livestock with the Mean Species Abundance biodiversity indicator to quantify the biodiversity impact of livestock through land use. Results showed weaker synergies and more trade-offs between environmental criteria in grassland than in mixed production systems. Efficiency in the utilization of feed and their associated land use is likely to drive the synergies in mixed production systems. Grassland systems based on extensive feed land use with high biodiversity values may have contrasted GHG emissions performances. Our global mapping of the relationships between environmental criteria could be used for spatially targeting decisions and actions.

Keywords: Environmental sustainability, Life Cycle Assessment, multi-criteria, intensity

1. Introduction

Livestock production faces a challenge in satisfying an increasing food demand while improving its environmental sustainability. In the next decades, global population growth, urbanization and higher incomes will lead to a 70% growth in the demand for animal products (FAO 2011). On a global scale, livestock are major contributors to food security but also to environmental impacts such as greenhouse gas (GHG) emissions (Gerber et al. 2013), water pollution (Carpenter et al. 1998) or biodiversity loss (Steinfeld et al. 2006). Large scale assessments of the environmental impacts of livestock reveal both hotspots of impact and sustainable systems; they are thus key to effective action and improvement.

Most of the recent global assessments of the livestock environmental performances focused on GHG emissions. They provided consistent results for the emission associated with different types of livestock products and for the hotspots along the supply chain (e.g., review in De Vries and De Boer 2010). Relying on these assessments, technical (Smith et al. 2008; Garnett 2009) and policy (Gerber et al. 2010; Steinfeld and Gerber 2010) options have been proposed to mitigate the contribution of livestock to climate change. Yet, environmental impacts of livestock production are not restricted to GHG emissions and carbon footprint cannot be used as an indicator of the overall environmental impact (e.g., for meat production, Röös et al. 2013).

In particular, livestock have a very strong impact on biodiversity. The main global driver of biodiversity loss is habitat change (MEA 2005; Foley et al. 2005). As major users of land resources, livestock have a strong contribution to this driver. About 30% of the global area is currently dedicated to livestock production through pastures and feed crops (Ramankutty et al. 2008). The Amazonian forest may host up to a quarter of the world's terrestrial species (Dirzo and Raven 2003). Its conversion to pastures (representing 85% of the new agricultural lands, Steinfeld et al. 2006) and soybean crops is an important threat to biodiversity. In Europe, grassland intensification and conversion to cropland have caused an important decline of farmland species (Vickery et al. 2001; Firbank et al. 2008). Overgrazing is an important factor causing habitat degradation through desertification and woody encroachment in arid rangeland system, which leads to decrease in the species richness of plant communities (Milton and Dean 1995; Asner et al. 2004).

Although there is large evidence of the global impact of livestock on biodiversity, very few quantifications exist and including biodiversity impacts in LCAs is still an emerging area of work. A framework (Mila i Canals et al. 2007; Koellner et al. 2013) and several characterization factors (review in Curran et al. 2011) have been proposed to compute biodiversity impacts through land use in LCAs. Most of the characterization factors are available at country to region scale (Koellner and Scholz 2008; Schmidt 2008; Goedkoop et al. 2012). At global scale, Koellner et al. (2013) proposed a standardized land use classification for computing biodiversity characterization factors (as part of the UNEP-SETAC Life Cycle Initiative). De Baan et al. (2013) relied on this classification to quantify the land use impact on biodiversity with a Biodiversity Damage Potential characterization factor. This study did not describe several levels of intensity within the pasture/meadow and annual crop land use classes, which is a limitation for quantifying the impact of livestock production with precision. Alkemade et al. (2009) developed a Mean Species Abundance (MSA) indicator and computed its value at global scale for various land use and intensity classes. Authors did not used the MSA within an LCA perspective but to predict the effect of global socio-economic scenarios on biodiversity, certain scenarios specifically addressing livestock production (Westhoek et al. 2011; Alkemade et al. 2012).

Most large scale environmental assessments have focused on one environmental criteria, and chiefly on GHG emissions. Both synergies and trade-offs are however likely to exist between the performances on GHG emissions and on other environmental impact categories, such as biodiversity. For instance, grassland systems often involve lower feed digestibility and thus higher enteric CH4 emissions (Eckard et al. 2010). However, in some regions grassland systems are crucial for maintaining rich biodiversity habitats, and both intensification and abandonment lead to the loss of a unique pool of species (Bignal and McCracken 2000). Quantifying both GHG emissions and biodiversity impacts is important to reveal how policy options targeting one criteria will involve benefits or conflicts with the other criteria.

The objective of this paper was to compare the GHG emissions and biodiversity impact of livestock production on a global scale. We combined the GLEAM model (Global Livestock Environmental Assessment Model, Gerber et al. 2013; Opio et al. 2013) which computes the global GHG emissions of livestock, with the MSA methodology in order to quantify the global biodiversity impact of livestock through land use. We investigated the relationship between performances on these two environmental criteria among commodities, production systems, and across scales.

2. Methods

2.1. Overview

The methodology was based on GLEAM which models global livestock supply chains in details and computes the GHG emission (Gerber et al. 2013, Section 2.2.). Computing the land use for feed is an intermediary output of the model (Figure 1). We used this intermediary output to develop a new component of GLEAM, which estimated the impact of livestock on biodiversity through land use. For this biodiversity component, we relied on the MSA methodology which provides a biodiversity value (expressed as Mean Species Abundance) for several classes of land use and intensity (Alkemade et al. 2009; 2012, Section 2.3.).

All computations addressed the global scale and were based on GIS raster layers with a resolution of 3 arc minutes (5*5km at the equator). The year of reference was 2005. We focused on a single species (cattle) but we described two commodities (milk and meat) and two production systems (grassland and mixed). For the dairy herd producing meat as a co product, no allocation was performed; environmental impacts (GHG emissions and MSA impact) were expressed by kg of proteins, summing proteins from milk and from meat.

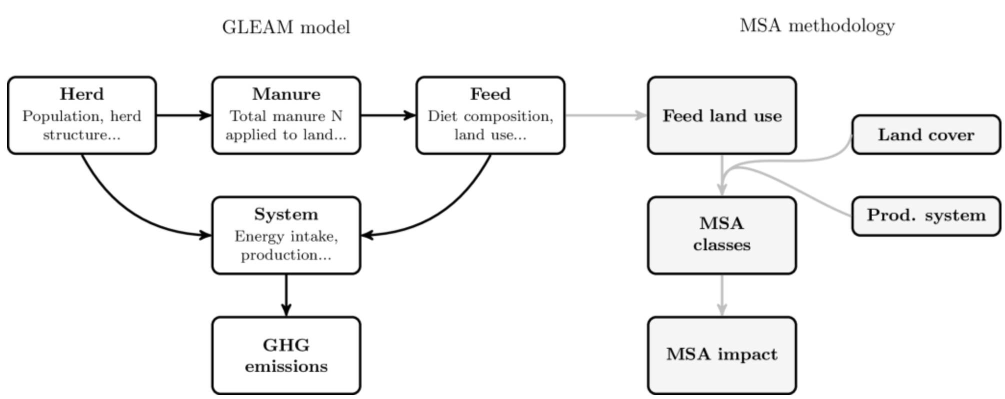


Figure 1. Overview of the modeling procedure used to compute GHG emissions, through the Global Livestock Environmental Assessment Model (GLEAM, Gerber et al. 2013) and the biodiversity impact, through the Mean Species Abundance methodology (MSA, Alkemade et al. 2009; 2012). Prod. = production. Adapted from Gerber et al. 2013.

2.2. GHG emissions: the GLEAM model

GLEAM is a novel modeling framework that enables a comprehensive analysis of the emissions of global livestock production (Gerber et al. 2013). It provides disaggregated estimates of the GHG emissions for the main commodities, production systems and world regions. The main GHGs in the agriculture context – CH_4, N_2O and CO_2 – are accounted for. GLEAM is built on modules reproducing the main elements of livestock supply chain: the herd module, the manure module, the feed module and the system module (Figure 1).

The GHG emissions results included emissions from feed production (main sources of emissions: N_2O from fertilization; CO_2, N_2O and CH_4 from energy use, fertilizer manufacture and land use change related to soybean cultivation) and livestock production (main sources of emissions: CH_4 from enteric fermentation and manure management; N_2O from manure management; CO_2 from on-farm energy use for livestock). For comparison with biodiversity impact through land use, we excluded emissions from manufacture of on-farm building and equipment, and post farmgate emissions. For a detailed description of the GLEAM model, refer to Gerber et al. (2013) and Opio et al. (2013).

2.3. Biodiversity impact: the MSA methodology

In order to compute the MSA values of different land use and intensity classes, Alkemade et al. (2009; 2012) conducted a meta-analysis and selected articles that presented data on species composition in disturbed (occupied) vs undisturbed (reference) land uses. No selection of specific species groups was performed; studies included in the meta-analysis addressed both plants and animals (mainly birds, mammals and insects). For each species k within each occupied land use i, the ration $R_{i,k}$ was calculated as:

$$R_{i,k} = \begin{cases} \dfrac{n_{i,k}}{n_{ref,k}} & if\ n_{i,k} < n_{ref,k} \\ 1, & otherwise \end{cases} \qquad \text{Eq. 1}$$

where $n_{i,k}$ is the abundance of the species k in an occupied land use i and $n_{ref,k}$ its abundance in the reference land use. The MSA of any occupied land use MSA_i is then calculated by summing and weighting the ratios $R_{i,k}$ of each species:

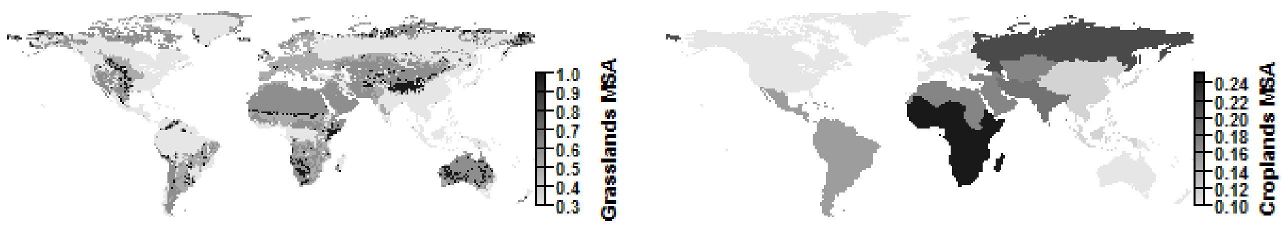
$$MSA_i = \frac{\sum_k \left(R_{i,k}/V_{i,k}\right)}{\sum_k 1/V_{i,k}} \qquad \text{Eq. 2}$$

where $V_{i,k}$ is the variance of the ratios of species abundances for each study and copes for differences between studies. MSA values vary between 0 and 1. MSA = 1 in undisturbed ecosystems where 100% of the original species abundances remains, conversely, MSA = 0 in a destroyed ecosystem with no original species left.

Table 1. Mean Species Abundance (MSA) value of the different land use and intensity classes of rangelands/grasslands, and croplands (Alkemade et al. 2009; 2012).

Land use and intensity classes	MSA value
Rangelands/grasslands	
Natural rangelands	1
Moderately used rangelands	0.6
Intensively used rangelands	0.5
Man-made grasslands	0.3
Croplands	
Low input agriculture	0.3
Intensive agriculture	0.1

Table 1 shows the MSA values of the different land use and intensity classes. Feed land uses as computed by the GLEAM model were translated into MSA land use classes (Figure 1). For each grid cell, we allocated a rangeland/grassland class (Figure 2a), and a cropland class (Figure 2b), corresponding to MSA values. For distinguishing between the different rangeland/grassland classes, we used information from three different layers mapping potential vegetation (ecoregions, Olson et al. 2001); global land cover (GLC 2000) and the distribution of grassland vs. mixed production systems (Gerber et al. 2013). We used the following hierarchic rules. The man-made grasslands class (MSA = 0.3) was allocated to grid cells with forest as potential vegetation, except for Europe where although forest is the potential vegetation, grasslands are sufficiently old to host specifically adapted species and very high biodiversity levels (Bignal and McCracken 1996; Benton et al. 2002). The natural rangelands class (MSA = 1) was allocated to grid cells with herbaceous land cover and grassland production systems. For grid cells with non herbaceous land cover (e.g., crops, crops/grass mosaic), the moderately used rangelands class (MSA = 0.6) was allocated to grassland production systems while the intensively used rangelands class (MSA = 0.5) was allocated to mixed production systems.



(a) (b)

Figure 1. Mean Species Abundance (MSA) value attributed to (a) the grasslands and (b) the croplands of each grid cell.

We further computed a characterization factor illustrating the impact of livestock production on MSA in each grid cell of the global raster:

$$MSA\,impact = \sum_i (1 - MSA_i) \times Area_i \qquad \text{Eq. 3}$$

where *(1-MSA$_i$)* stands for the loss of MSA in land use *i* compared to the reference land use. For instance, an MSA value of 0.6 in extensive grasslands means that the MSA loss *(1-MSA$_i$)* is 0.4, i.e. that 40% of the mean species abundance is lost compared to the reference land use. *Area$_i$* is the area of land use class *i* necessary to produce the feed consumed by cattle in the grid cell. Therefore, the MSA impact of land use for feed is not allocated where feed is produced but where it is consumed. For instance, the MSA impact of soybean cultivated in brazil and consumed by cattle in a given European grid cell will be allocated in this European grid cell. As a consequence, livestock production in a given grid cell could have a land use impact on a larger area than the grid cell area. The MSA impact is expressed as an *MSA loss * km^2* and then divided by the kg of protein produced.

2.4. Spatial analyses

We computed average GHG emissions and MSA impacts across grid cells, at the level of agro-ecological zones (intersection of climate zones and global regions, Fischer2008) and climate zones (arid, humid, temperate).

We investigated the relationship between GHG emissions and MSA impact environmental criteria at local (grid cell) scale. For the two environmental criteria, the environmental impact of a grid cell was compared with the average impact at sub-regional level (moving window average). If the impact on the two environmental criteria were both lower or both higher than the sub-regional average, we considered that there was a synergy between criteria for the grid cell. Conversely, if the impact was higher than the regional average for one environmental criteria and lower for the other, we considered that there was a trade-off between environmental criteria for the grid cell.

All analyses were performed using the R software, version 3.0.3 (R Core Team 2014). We used the raster package (Hijmans 2014) to perform GIS analyses.

3. Results

3.1. Relationship between environmental criteria across agro-ecological zones

As a general trend across agro-ecological zones, there was a synergy between the climate change and biodiversity performances: agro-ecological zones with lower GHG emissions also tended to have lower MSA impacts (Figure 3). This general trend was observed similarly for dairy (Figure 3a) and beef (Figure 3b) cattle, and for grassland and mixed production systems. For both commodities, the correlation between GHG emissions and MSA impact was much lower in grassland production systems (R^2 = 0.2936 and 0.2295 for dairy and beef cattle, respectively) than in mixed production systems (R^2 = 0.8577 and 0.7658 for dairy and beef cattle, respectively).

Figure 3 also shows that environmental impact on the two criteria was higher for beef cattle than for dairy cattle. For the two commodities, grassland production systems had a slightly higher MSA impact than mixed production system. In terms of GHG emissions, the difference between production systems was very small.

Environmental impacts on the two criteria across climate zones were similar for the two commodities (results not show). Environmental impacts were the highest in arid regions and the lowest in temperate regions. Humid regions had GHG emissions levels close to those of arid regions while they had an MSA impact intermediate between those of arid and temperate regions.

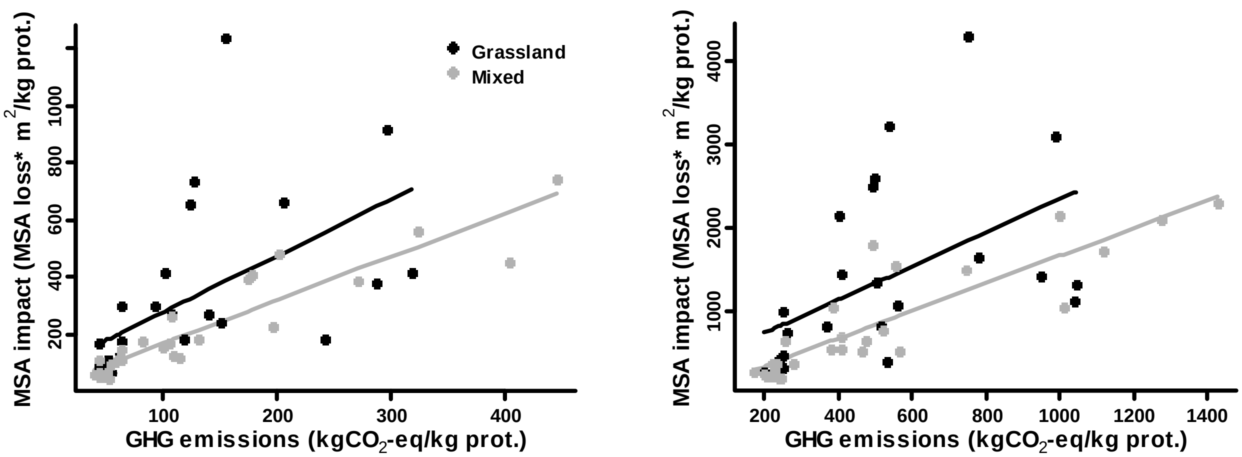


Figure 3. Relationship between GHG emissions and MSA impact per unit of production across agro-ecological zones. (a) Dairy cattle; (b) Beef cattle. Each point stands for the mean GHG emissions and MSA impact of one production system (see legend), within one agro-ecological zone. Regression lines are plotted. Prot. = protein.

3.2. Relationship between environmental criteria at local scale

Figure 4 shows the global distribution of synergies and trade-offs between GHG emissions and MSA impact. For both commodities, regions with a higher concentration of trade-offs included brazil and the western part of North America, as well as India for dairy production and western Europe for beef production. The rest of Europe Asia, and Africa showed more uniform proportions between synergies and trade-offs. Trade-offs tended to be more frequent in arid climate, grassland production systems and areas with high MSA values while synergies tended to be more frequent in temperate systems and area with higher yields.

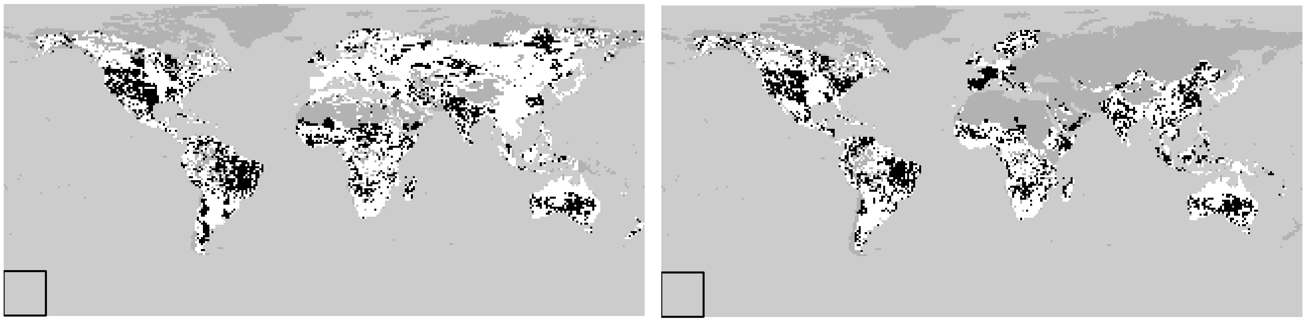


Figure 4. Relationship between GHG emissions and MSA impact per unit of production at local (grid cell) scale. (a) Dairy cattle; (b) Beef cattle. White = impact on the two environmental criteria are both lower or both higher than the sub-regional average (synergy). Black = impact is higher than the regional average for one environmental criteria and lower for the other (trade-off). Grey = no production. Black square = size of the moving window for the sub-regional averages.

4. Discussion

4.1. The MSA indicator

The MSA indicator is one of the very few biodiversity characterization factors for land use available at global scale. One limitation of the MSA indicator is that it does not make it possible to account for potential differences in conservation value, at the species and at the ecosystem levels. At the species levels, all species groups were included in the MSA. The MSA indicator is thus expected to be a good proxy for the overall biodiversity, although it is based on published literature where certain taxa are underrepresented (e.g., arthropods). However, common species and patrimonial or red listed species have the same contribution to the MSA. The IUCN red list is a widely recognized system for classifying species according to their risk of extinction and biodiversity indicators based on the red list are a useful tool for targeting conservation actions (Butchart et al. 2004). Characterization factors can be computed separately for common and red-listed species (e.g., in Koellner and Scholz 2008 for central Europe) but this distinction is not made in the MSA. Besides, while most biodiversity characterization factors are based on species richness (e.g., Koellner and Scholz 2008; Michelsen 2008; Schmidt 2008; Goedkoop et al. 2012), the MSA is based on species abundance which does not capture information about species extinction.

At the ecosystem level, the MSA value of each land use and intensity class is global and does not account for regional differences. It means that the biodiversity value of undisturbed forest – or the biodiversity loss following its conversion to pasture – is the same in Siberia and Amazonia for example. Yet, it is recognized that forests in certain specific areas are biodiversity hotspots (e.g., Amazonia, Dirzo and Raven 2003). The framework developed by the UNEP-SETAC Life Cycle Initiative (Koellner et al. 2013) includes a structure for including such regional differences, i.e. for the regionalization of land use elementary flows. Characterization factors computed by De Baan et al. (2013) follow this regionalization structure. The MSA indicator includes more precise land use and intensity classes, specifically adapted to livestock production. However, the absence of regionalization is a limitation. In the context of livestock production, the biodiversity value of grazing lands of varying intensity is very likely to differ between global regions. We accounted for one of these difference by making the assumption of extra MSA value to European grassland (compared to Alkemade et al. 2009; 2012) because they are very old ecosystems, although located in a forest ecoregion. Other differences were not considered; for instance, the management intensity threshold leading to rangeland degradation is lower in humid and arid regions than in temperate regions (Asner et al. 2004). Using information on rangeland productivity and livestock density would be an interesting development to regionalize the MSA values of grasslands and rangelands.

The UNEP-SETAC Life Cycle initiative recognizes that the effects of land use on biodiversity can be divided in three main stages that occur successively over time: land transformation, land occupation and land restoration (Lindeijer 2000; Mila i Canals et al. 2007). In this study, we only focused on occupation impacts. Calculating transformation impacts (e.g., impacts from land use change) and permanent impacts (i.e. impacts that cannot be recovered even after restoration) requires region and ecosystem specific data on the time and success of restoration (De Baan et al. 2013). Such data was not available; moreover the GLEAM model that we used to compute land occupation is a static model and its input datasets are not available as time series.

4.2. Impact categories coverage

The MEA (2005) recognizes five main driver of biodiversity loss at global scale: habitat change, climate change, pollution, invasive species and overexploitation. On a global scale, livestock production contributes directly or indirectly to each of these five drivers (Steinfeld et al. 2006). We focused on the habitat change driver and described it through land use only, which does not cover other of its components such as spatial heterogeneity and habitat fragmentation. This focus on land use leads our results to underestimate the overall biodiversity impacts. It also overemphasizes the role of productivity: productivity gains at feed production and animal level both strongly drive the calculated MSA impact. This results in a bias in favor of high productivity systems. Land use is expected to be the main driver of impact on biodiversity for extensive grassland systems which use large area to generate one unit of product because the conversion of grass to animal protein is rather inefficient (Wirsenius et al. 2011). Mixed systems and intensive grassland systems show higher yields, they need less area which limits their impact on biodiversity through land use, despite lower MSA values. However, they often involve important pollution that causes significant biodiversity impacts which are not captured in our results. This pollution includes two main categories. The first one is nutrient pollution which is associated with animal concentration (Peyraud et al. 2012) and can lead to biodiversity loss through acidification and eutrophication in soils and water (Carpenter et al. 1998). The second one is associated with higher input intensity; it is the release of ecotoxic components in the environment, mainly pesticides (at the feed production stage) and veterinary products (including hormones, at the animal husbandry stage). Hormonally active pesticides cause adverse effects on a wide range of organisms (Colborn *et al.*, 1993) and have recently been pointed as one of the responsible of bee population decline (vanEngelsdorp and Meixner 2010). Several veterinary products used on livestock have also been shown to impact biodiversity, such as anti-inflammatory drugs (Baillie 2004), hormones (Soto et al. 2004) or anthelmintics (Lumaret and Errouissi 2002).

Our results did not show significant differences in GHG emissions between grassland and mixed production systems. Adding the impact of climate change on biodiversity should not lead to important changes in the relative impact of grassland vs. mixed production systems on biodiversity. However, it could reveal new synergies between the two environmental criteria.

Many LCA studies on livestock focused on climate change as a single environmental criteria (see review in De Vries and De Boer IJM 2010). Adding land use impacts on biodiversity to the climate change impact is still far from a complete coverage of all the categories relevant to the environmental performance of livestock production at global scale. Interestingly, most of these additional categories are midpoint impacts which also have an effect on biodiversity. Including their assessment and their effect on biodiversity would be an important further development of our approach. Characterization factors to model their effect on biodiversity in LCA at global scale are still rare. In the MSA methodology, biodiversity characterization factors are also available for the impact of fragmentation and climate change (Alkemade et al. 2009). Other global characterization factors exist for climate change (Schryver et al. 2009), water use (Pfister et al. 2009) and ecotoxicity (Rosenbaum et al. 2008) while country to region characterization factors exist for eutrophication and acidification (Van Zelm et al. 2007; Struijs et al. 2011).

4.3. Implications

Across agro-ecological zones and at local scale, there was a weaker correlation and more trade-offs between GHG emissions and MSA impact in grassland production systems than in mixed production systems. In mixed production systems, feed mainly come from intensive land uses with low MSA values. Efficiency in the utilization of feed is a way to improve both environmental criteria. It makes it possible to use less area of intensive feed land uses with low MSA values, which also decreases GHG emissions associated with feed cultivation. At the same time, production is increased which leads to lower environmental impact per unit of product. In grassland production systems however, another option than efficiency to mitigate biodiversity impacts is to use more extensive feed land uses with high MSA values. This option can have contrasted effects on GHG emissions performances because more extensive systems have lower production levels and can be associated with higher emissions from manure deposition and enteric fermentation (Gerber et al. 2011).

By mapping synergies and trade-off between two environmental criteria on a global scale and at fine resolution, our method could provide a useful tool for spatially targeting interventions, or further investigations of the farming system properties. We reveal different relationships between the GHG emissions and biodiversity performances. The systems where performances on both criteria are higher in the surrounding region could be studied in order to apply beneficial management options to other systems, e.g., to those were performances on both criteria are lower than those of the neighboring systems. Systems were only one environmental criteria performs better than the surrounding region could reveal trade-offs between criteria that needs to be considered when designing interventions. Limitations of the implications of our results at the grid cell level include that certain parameters of the model only have a country resolution (e.g., animal's ration in OECD countries), and that management decisions could have different constraints across grid cells, even within a sub-region.

5. Conclusion

This study provides a tentative global quantitative assessment of the environmental performances of livestock production on two criteria: GHG emissions and biodiversity. It is a first attempt to develop multi-criteria assessments over such large scale, which are key to inform decision and action towards improving the overall sustainability of the livestock sector. Our preliminary results show that both synergies and trade-offs exist between the performances on the GHG emissions and biodiversity criteria. With our approach, more frequent and stronger synergies were found in mixed production systems where efficiency (i.e. decreasing the feed land use area while increasing production) could be a way to improve performances on both GHG emissions and biodiversity criteria. Weaker synergies and more trade offs were found in grassland production systems. Further developments, and testing of our approach are however required before results can be used for decision making. Improvements would include the inclusion of additional and regionalized biodiversity impacts.

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Optimizing the LCA data processing for food products in the context of Life Cycle Sustainability Assessment: challenges and opportunities

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ABSTRACT

Life Cycle Inventory (LCI) is the most time consuming and cost intensive part of a Life Cycle Assessment (LCA) study. This is especially the case for Life Cycle Assessment (LCA) of food products, where the numerous underlying agricultural systems represent generally a major part of the environmental impacts and a considerable amount of data to be collected. In particular the modelling of direct field and farm emissions requires a large amount of parameters (e.g. field management, soil properties, climatic conditions etc.) and involves iterative calculation loops at several levels (crop, field, farm and product). Through the generic tools SALCA*crop* and SALCA*farm*, an efficient and modular solution in the frame of Swiss Agricultural Life Cycle Assessment (SALCA) is already existing. Most of the LCA projects for field, farm or food LCA can be described in a workflow including SALCA*farm* and SALCA*crop*, but some adaptations are necessary according to the goal and scope of the project under study. In addition, if the data provider is not a LCA practitioner or a specialist in agriculture, a data collection tool must be developed. This was the case in the CANTOGETHER (Crops and ANimals TOGETHER) European project aiming at the environmental assessment of case-study farms and regions. Conventional LCA software tools such as Simapro 8, but also specific modeling tools such as *SALCAfarm*, *SALCA-BD*, *SPACSYS*, *RUSLE2*, and the *SIKtool-EFM* were integrated. To minimize the time-consumption of such adaptations and developments, a broader IT solution that can manage a high variety of LCA projects must be established in the future. Based on the experience of *SALCAcrop*, *SALCAfarm*, CANTOGETHER, and other LCA studies of food products, such a solution is under development: *SALCAfuture*. The main components will be a user-friendly data collection website, a centralized calculation and administration tool, and an analysis and assessment tool. SALCA*future* will be able to integrate a wide portfolio of LCA and eventually Life Cycle Sustainability Assessment (LCSA) projects from the agrifood sector. The various components will also be accessible to external users.

Keywords: farm and food LCA, direct emissions LCI, data collection, LCA tools

1. Introduction

The Life Cycle Inventory (LCI) phase is generally the most time consuming phase of a Life Cycle Assessment (LCA) study. In food LCA, the numerous and diverse underlying agricultural production systems contribute generally significantly to the environmental impacts (Weidema and Meeusen 2000). To estimate these impacts, the indirect emissions from the technosphere, but also the field and farm direct emissions must be modelled. This requires a broad and large amount of data: field management and related product use, soil properties, climatic conditions, animal and manure management, location, infrastructure, machinery, etc. This data is present at different levels from the crop, the field to the farm level. For food LCAs the impacts must be allocated to the product level, which requires additional data, for example the ratio of internal use and the amount of products that are bought or sold[1].

Concerning the calculations, the complexity is also present since several stages of iteration are necessary according to the crop, field, and farm and product level. In addition to this, the use of linear and non-linear models, and the high amount of data leads to the requirement of an efficient calculation procedure. This is achieved by SALCA*crop* and SALCA*farm,* generic tools for field or farm and product LCA.

Since their release, these tools were used in a considerable number of projects, and some experience could be gained regarding their future development. In addition, the recent discussion and development of conventional LCA in the direction of Life Cycle Sustainability Assessment (LCSA) establish the need for tools that can incorporate new sustainability metrics (Guinée et al. 2011). The goal of this paper is to describe the concept of a next-generation of tools for field or farm and product LCAs. Firstly, the current workflow of SALCA*farm* and SALCA*crop* will be briefly described with the perspective of a typical LCA project. Secondly, the workflow developed specifically for the project Crops and ANimals TOGETHER (CANTOGETHER 2014), including an adaptation of SALCA*farm* and the development of a flexible data collection tool will be presented. Finally the workflow of the next-generation of tools, in the frame of the project SALCA*future*, will be introduced.

[1] In this paper a product can be anything that the farm produces: a food product, but for example also energy from a bioenergy facility. C*rop level*, do not refers only to the crop product itself, but also to the calculation that requires data at the detail level related to the crop.

2. The workflow of SALCA*farm* and SALCA*crop*

The current tools used for field or farm and product LCAs are SALCA*crop* and SALCA*farm*. These generic tools were already described in detail in the reference Nemecek et al. 2010, in particular regarding the emission methods, the calculation procedure, and the impact assessment methods. In this paper we focus on the overall workflow with the perspective of a typical LCA project.

SALCAcrop and SALCAfarm are currently used for each project in the workflow of Figure 1, following the conventional phases of a LCA project. Firstly a data collection tool, generally based on Excel is used, or the data is entered directly in the input files PIfarm or PIcrop by LCA practitioners or agriculture specialists. This results in the agricultural Production Inventory (PI). In case a data collection tool exists, it was developed according to the specific goal & scope of the project. The development of such a tool can be intensive in terms of resources, in particular if the users aimed are for example farmers, and the required flexibility is high.

The next step is the calculation of the intermediate inventory, containing direct emissions, and inputs and outputs from technosphere. This is done with either SALCAfarm or SALCAcrop and results in an ecospold1 file that is imported in the conventional LCA software SimaPro7. It involves linear and non-linear calculations at the crop, field, and farm and product level with several direct emission calculation modules. The treatment of numerous farms is possible (batch processing). For most of the projects some adaptations are necessary, also according to the goal and scope of the study, for example the adaptation of national emission factors or the implementation of country-specific product commercial names.

Finally the results are imported and analyzed in SimaPro7 and extracted generally in MS Office tools for further assessment, analysis, and reporting. This includes several manual steps, which are reduced by generating one ecospold1 file with several farms in the case of a batch calculation procedure.

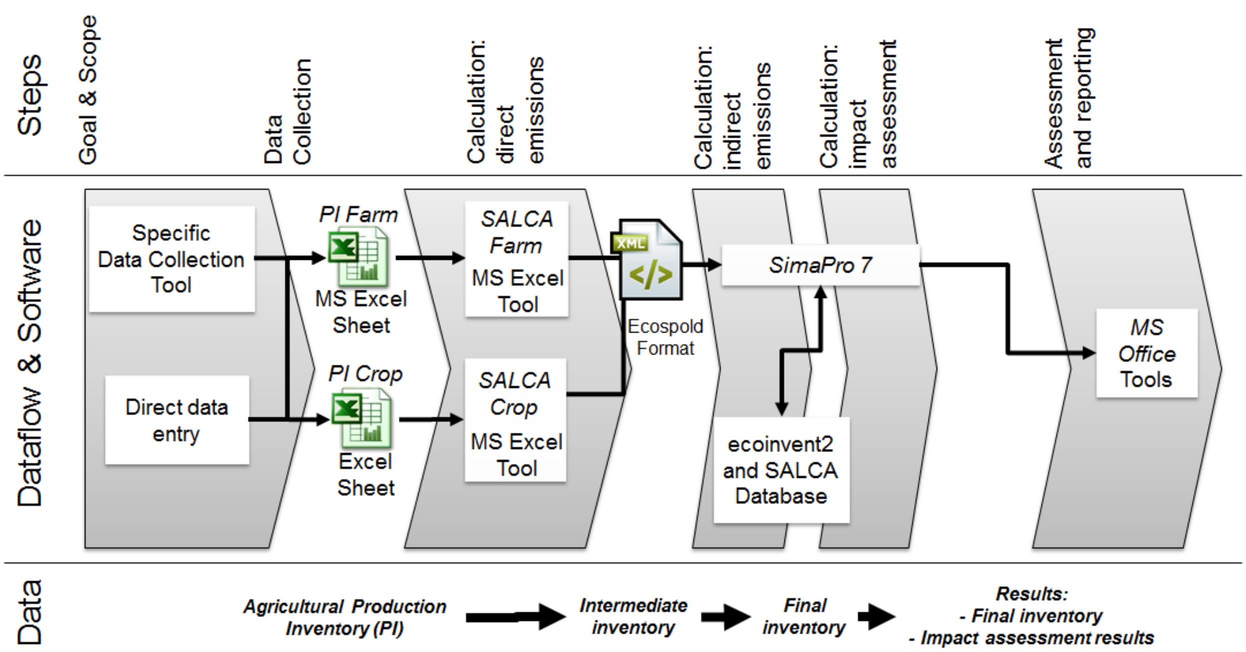

Figure 1: Current workflow, dataflow, and tools, according to the steps of a typical agricultural or food LCA project with SALCA*farm* and SALCA*crop*.

In this general workflow, the main strengths of SALCA*crop* and SALCA*farm* tools are:

- high scientific quality
- reliability, automation, and efficiency of the calculation of direct emissions at crop, field, and farm and product level
- modular construction of emission models (models parameters and calculation can be rapidly adapted to the goal and scope)
- possibility to perform batch calculation of several farms
- based on a conventional software (Excel) that is understood by a important share of the LCA practitioners or agriculture specialists, and that allows quick adaptations.

In practice, an important amount of projects requires the development of a specific data collection tool. In addition, although the modular construction of SALCA*crop* and SALCA*farm* allows some rapid adaptations of the models, this is not the case anymore if the adaptations change the input structure of the modules (e.g. adding new inputs, or increasing the detail level of the inputs). In summary theses developments and adaptations are time-intensive and may increase the resources needed to perform field or farm and product LCAs, depending on the goal and scope of the LCA project considered. In the next section we will illustrate theses adaptations by taking the example of the CANTOGETHER project.

3. The workflow of CANTOGETHER

In the CANTOGETHER project, a goal and scope was set with the assessment of 10 case-studies for 17 products and 7 scenarios. The case studies were made of either conventional farms, experimental farms, or pool of farms at regional level. Some case studies were containing an important number of fields, with a maximal number of about 45 fields. The data providers were agriculture specialists but could also be farmers in some cases. In this context, a data collection tool with demanding requirements for flexibility and user-friendliness had to be developed.

In addition, the methodological choices that were made for CANTOGETHER implied the use of various tools, including conventional LCA software tools such as Simapro 8, but also specific modeling tools such as *SALCAfarm*, *SALCA-BD*, *SPACSYS*, *RUSLE2*, and *SIKtool-EFM* (Cederberg et al. 2013). This resulted in the workflow described in Figure 2, with the following elements:

- **CANCollect:** Excel-based data collection tool. A high flexibility is available, with for example the possibility to add pesticides, mineral and organic fertilizers with the corresponding properties (e.g. N, P2O5, K2O content) and the possibility to describe field management events with a high level of detail (e.g. including the date of the event). The inputs of each modelling tools were analyzed in-depth and synergies were exploited in order to reduce the amount of data to be collected.
- **CANCalc:** Excel-based tool that contains two modules:

  1) The data transmission module, which extracts the data from CANCollect and implements it in each calculation tool (adapted SALCA*farm*, SIKtool-EFM, RUSLE2, SPACSYS, SALCA-BD). This includes some sorting, mapping, and database queries. In other words the data is translated from a user-friendly perspective to a software-friendly perspective.

  2) An adapted version of SALCA*farm* that controls the calculations and extracts the outputs of the SIKtool-EFM, SPACSYS and RUSLE2, and integrates new inputs and methods (e.g. for the water stress index). The inputs needed are at crop, field and farm level.

- **SIKtool-EFM:** Excel-based tool that calculates the livestock enteric fermentation and manure management emissions. The inputs needed are mainly at farm level (Berglund and Cederberg 2014).

- **SPACSYS:** C++ based stand-alone tool with inputs and outputs of all components organized as a database in either Microsoft SQL Server 2000, Access 2000 or MySQL5.0. This tool simulate root systems, nitrogen cycling, phosphorous cycling, water flows, plant growth and direct emissions (Wu 2013). The inputs needed are mainly at crop and field level, including daily data for weather and field management.
- **RUSLE2:** C++/SQLite based stand-alone tool that is used to compute the erosion caused by rainfall and its associated overland flow (USDA 2008). The inputs needed are mainly at crop and field level, including daily data for weather and field management.
- **SALCA-BD:** Java and Excel based tool that assess the impact of agricultural land use on biodiversity (Jeanneret et al. 2008). The inputs needed are mainly at crop and field level, including monthly data for field management.
- **CANAnalyse:** Excel-based tool that extracts the results of the various tools involved and allows in-depth analyzes.

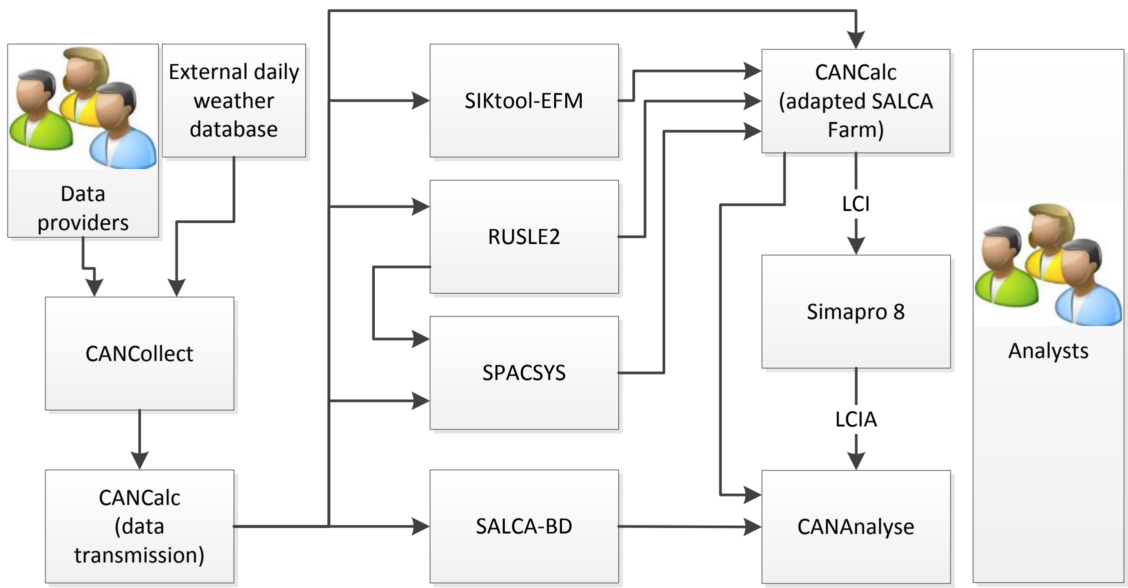

Figure 2: Workflow of the CANTOGETHER project, including the data collection tool CANCollect, the calculation tools (CANCalc, SIKtool-EFM, RUSLE2, SPACSYS, SALCA-BD and Simapro8) and the analysis and assessment tool CANAnalyse.

Most of the impact assessment results are calculated with the help of a traditional LCA software based on the inventory extracted from CANCalc. However, some tools deliver directly results at impact assessment level to CANAnalyse. For example, CANCalc provide the Water Stress Index (Pfister et al. 2009) for the water consumed at field level, and SALCA-BD provides a biodiversity assessment.

In summary, the development of the data collection tool CANCollect and the calculation tool CANCalc were resource intensive for the following reasons:

- Overall complex workflow for the data processing, with a high variety of links between the various tools
- Requirement of a user friendly data collection tool that can be transmitted to farmers and that allows to enter all the necessary data for the tools
- The presence of at least one crop or field level modelling tool with daily data requirements
- The presence of several stand-alone tools with their own lists of input data (i.e. their own terminology) and automation possibility
- The very large amount of data coupled with the presence of stand-alone tools: the overall solution must be automatized as far as possible.

Based on this experience of the CANTOGETHER project, the next generation of SALCA*farm* and SAL-CA*crop* should allow more extensive adaptations and avoid the development of a specific data collection tool for each LCA project.

4. The workflow of SALCA*future*

Projects in the field of agricultural or food LCA may have a high variety of goals and scopes which leads to different data collection needs, calculation models, and representation of results. As it was described in the previous sections, the tools SALCAcrop and SALCAfarm must be adapted and modified for most of the projects. In addition, the current excel input file is originally made for LCA practitioners and agriculture specialists and cannot be directly circulated to other stakeholders. This results in the development of specific data collection tools in most of the projects.

As described in the case of the CANTOGETHER project, some existing tools are already delivering some results at impact assessment level. This is due to the fact that some metrics are not integrated in traditional LCA software, because they are for example too specific to agricultural systems, or too recent. In the future, similar pathway would be necessary for new sustainability metrics that might not be supported in traditional LCA software (for example animal welfare, soil quality, social aspects etc.). Eventually this would allow integrating LCSA projects in the workflow presented before.

In this context, the project SALCAfuture was started with the aim of developing the next generation of SALCcrop and SALCA*farm*. The fundamental improvements are (Figure 3):

- A web-based application for the data collection, SALCA*collect*, with a high user-friendliness, and a data quality control. The data quality control will reduce the necessary number of data collection iterations.
- The merging of SALCA*crop* and SALCA*farm* in a centralized tool, SALCA*tools,* for the modelling of direct emissions. For the transparency and reproducibility of results, a version and user management will be implemented. In addition, it will be possible to import data based on .XML format
- The Implementation of Simapro 8 with the ecoinventV3 database
- A flexible analyzing and assessment tool (SALCA*analyse*)
- A high automation of the workflow
- A framework that allows the development of new sustainability metrics and eventually the integration of LCSA projects
- An improved accessibility for external user for SALCA*collect* but also for SALCA*tools* and SAL-CA*analyse.* This will improve the collaboration with partner in some projects, and also allow more transparency, allowing for example external reviews

In summary, SALCA*future* will be able to integrate a wide portfolio of LCA and eventually LCSA projects from the agrifood sector at an international scale. The web-application SALCA*collect* will, by definition, be easily circulated to any users, and will not require any installation of software. Typical users would be farmers, consultants in the agricultural sector, partners from others research institutes, or other specialists, and should be able to fill the data without any LCA knowledge. The modelling and calculation tool SALCA*tools*, and the analyzing and assessment tool SALCA*analyse* will be accessible to partners with LCA expertise, and will require the use of a widely available client software. The overall resources needed for the adaptations of the tools to a specific project will be reduced, allowing the data providers to enter data more efficiently and LCA practitioners to perform more in-depth assessments.

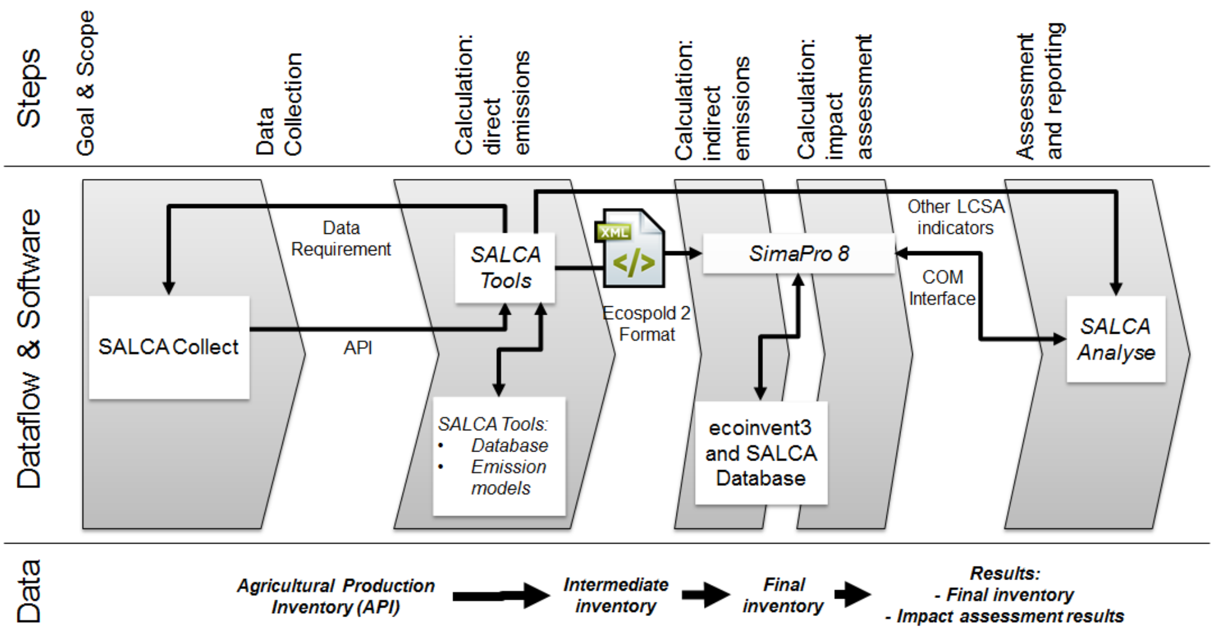

Figure 3: Workflow of SALCA*future* according to LCA Steps, dataflow, data, and the involved software.

6. Conclusion

In this paper, it was showed that current generic tools such as SALCA*crop* or SALCA*farm* can perform efficient calculation of Life Cycle Inventory (LCI) of agricultural systems. However, due to the very high variety of goal and scopes in LCA projects, a specific data collection tool must be developed, and some considerable adaptations of the existing tools must be achieved specifically for most of the projects.

This was typically the case for the project CANTOGETHER, with the development of a decentralized and flexible data collection tool, and with a vast adaptation of SALCAfarm including some integration of external stand-alone modelling tools.

Such developments and adaptations, specific for most of the projects, needs generally intensive resources and should be reduced with a higher flexibility of the tools. In addition, the specificity of agricultural systems and the emergence of Life Cycle Sustainability Assessment (LCSA) further increase the requirements in regards to the flexibility. To take these facts into account, the next generation of SALCA*crop* and SALCA*farm* was presented in the frame of the project SALCA*future*. In particular, SALCA*future* will handle these challenges and improve the availability of the tools for external users.

7. Acknowledgments

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Food service: climate issues and water demand of meals

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ABSTRACT

Food service companies are important stakeholders in promoting sustainably produced and healthy foods. We worked in a transdisciplinary team to calculate the greenhouse gas emissions and virtual water demands of vegetarian and non-vegetarian meals in Germany using the LCA method. Partners from different scientific disciplines and the manager of a food service company as out industrial partner collaborated on the study's design and the definition of goal and scope. The result of the study was a management plan for the incremental implementation of short and long-term choices based on environmental sustainability indicators. The major reduction potential rests in shifting eating habits from quantity to quality, specifically to high quality organic meat and in using renewable energy during food preparation. Linking LCA to transdisiplinary approaches is important for the development of environmentally responsible choices considering the food service company's internal preferences and regional infrastructural constraints.

Keywords: food service, sustainable foods, carbon footprint, virtual water demand, LCA

1. Introduction

Individual eating behaviors and attitudes have lasting effects on the development of sustainable lifestyles. Establishing balanced global dietary patterns will be a major challenge in the coming decades as growing affluence is a major driver behind the increasing demand for food (Tilman et al. 2011). There is an increasing tendency towards eating outside the home. Such is the case for school meals, canteen meals and catering companies offering their services at events. In 2012, the turnover of the German food sector was about 68 billion-euro and the trend is rising. Therefore, food service companies are important stakeholders in promoting sustainably produced and healthy foods. In Germany, the contribution of organic products in the food sector is estimated at 300 million-euro per year which is 0.5 % of the total market for organic food products (MKULNV 2010).

Over the past several years, different communication labels such as primaKlima, EMAS or Stop Climate Change were established to certify a climate friendly and sustainable performance of companies and effort were made to develop sustainability standards for restaurants (e.g. Baldwin et al. 2011). At first glance, the definition of system boundaries and functional units are often not recognized. In order to have a tool to transfer knowledge to consumers, especially in the food service sector, it makes more sense to relate environmental impacts to kilocalories than to kilograms of food. Most of the studies aiming to support sustainable decision making focus on the ecological perspective and underestimate the complexity of economic and social aspects (Binder et al. 2012).

In this study, we worked in a transdisciplinary setting in order to extend the research beyond the specialized knowledge of ecological dimensions of food provision. We valued this transdisciplinary approach as a means to fundamentally understand the situation of food service companies and discover solutions using problem-driven research. Despite a long tradition of application to questions of sustainable development, neither trans- nor interdisciplinary research approaches are strictly defined. The two approaches are also often applied simultaneously. In our understanding, a transdisciplinary approach combines and synthesizes scientific systems and societal systems (Hirsch-Hadorn et al. 2006).

Therefore, depending on specific aims, food service companies have several options for environmentally responsible choices regarding their services. We conducted our study using an LCA framework with the goal of identifying leverage points and priorities for reducing climate impacts and the water use for small and medium enterprises.

2. Methods

In our study, we cooperated with a food service company that runs a canteen kitchen and a catering service. Our industry partner has observed a rising demand amongst costumers for quality food and an interest in environmental impact labeling of the company’s foods. We worked in a mixed team with partners from different scientific backgrounds, including ecology, economics and nutritional advisers. In order to address problem-driven research questions, the manager of the food service company played an important part in the team, providing practical viewpoints that brought new insights to the scientific partners. Using this transdisciplinary approach, we collaborated with our partners to define specific questions to be answered over the course of the project.

The project’s starting workshop provided the platform for intensive discussions with all partners resulting in the definition of the goal and scope of the LCA, as well as the definition of the functional unit. Moreover, the definition of the system boundaries and samples for sensitivity analysis were settled. We first focused on climate impacts as the key indicator of environmental performance of two standard dishes served in the food service canteen. We choose a vegetarian and a non-vegetarian plate. Table 1 gives details on the different ingredients. One of the food service company’s typical dishes is couscous with mixed vegetables accompanied with a full-fat sour cream used as a flavoring ingredient. In order to evaluate the differences in environmental impact of beef and pork, we choose typical German meatballs with gravy and potatoes. The meatballs are composed of 50 % pork and 50 % beef meat. We also conducted an analysis of the water demand of both dishes, which we termed virtual water demand. The virtual water demand was calculated in order to evaluate potential tradeoffs between the two indicators.

Table 1. Isocaloric meal ingredients and baseline for the identification of reduction measures

Lunch plate	Ingredients	Energy content [kcal]	Contribution [%]
Vegetarian	Couscous	68	14
	Vegetable mix (corn, pepper etc.)	34	7
	Olive oil	21	4
	Sour cream	378	76
	Total	500	100
Meat	Meatballs	374	75
	Gravy	5	1
	Potatoes	121	24
	Total	500	100

The functional unit was one portion of a typical German vegetarian and non-vegetarian dish offered by the company. Considering the importance of nutrition to the consumers, we used the energy content as functional unit for representing the results. We evaluated two isocaloric diets, one vegetarian and one non-vegetarian dish à 500 kcal from “farm-to-fork” (Baldwin et al. 2011). The meal ingredients are presented in Table 1. GHGE (greenhouse gas emissions) and water use were analyzed from agricultural production and its upstream supply processes (e.g. animal feed, fertilizers) along the supply chain to the serving counter in the canteen regarding food processing steps (such as slaughter), storage, and food preparation in the canteen kitchen (see Figure 1). Primary data from the canteen’s most important suppliers and processing companies were collected via questionnaires and in interviews. The catering company provided data and energy use on food preparation and food wastes, as well as cooking and chilling techniques, and the re-heating of the meals in the canteen.

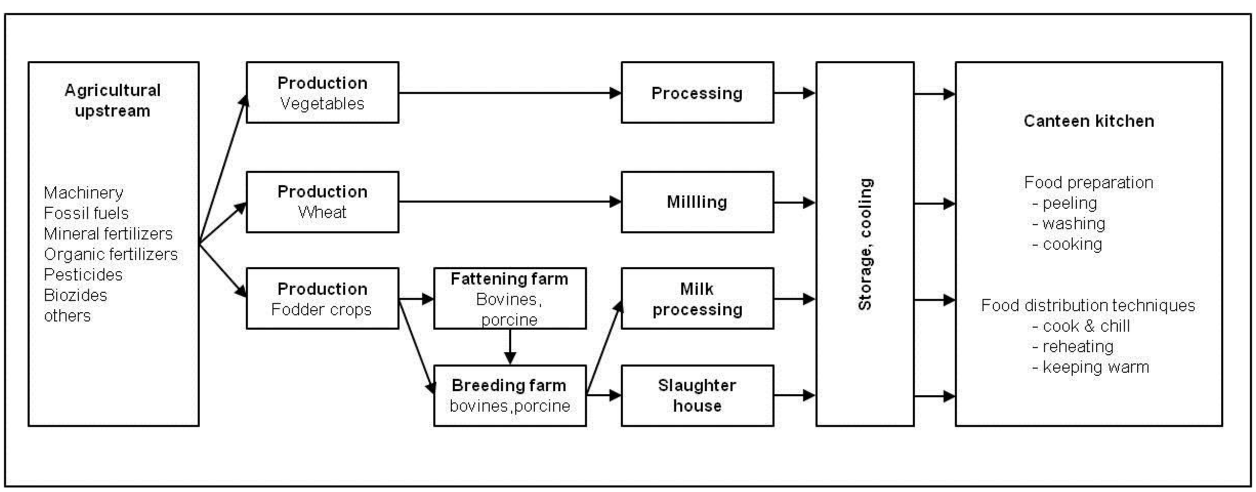

Figure 1. System boundaries for a typical lunch plate offered by the food service

2.1. Analysis of climate effects (GWP) and virtual water demand

This study used the life-cycle-assessment (LCA) method, following the ISO 14040 and 14044 standards (ISO 2006). The calculation of the carbon footprint was carried out using the SimaPro software, as well as the GEMIS 4.42 and ecoinvent v2.2 databases. The carbon footprint was calculated according to IPCC (2006). Primary data on agricultural production systems, specifically regarding yields, fertilizer input, cropping systems and supply chain, was available directly from the food service company's suppliers in Germany. Gaps in the data were filled with data from comparable agricultural production systems in similar climatic conditions in Austrian regions (Hörtenhuber et al. 2011, Lindenthal et al. 2010). Data on regional transport, including the different means of transport (truck size, refrigerated transport) and transport distance were obtained from questionnaires. Data on imported vegetables from South Spain, specifically on agricultural production, processing and means of transport, were obtained from Theurl et al. (2014). Food waste factors, obtained from primary data of the food service company for meat, fresh and frozen vegetables, were considered in the calculations of GHGEs and water use. The calculation of GHGEs is based on a specific, locally-adapted farm model in which methane emissions, as well as direct and indirect N_2O emissions are considered in the calculation of the GWP (Theurl et al. 2014; Lindenthal et al. 2010). Direct and indirect emissions due to land use changes from feedstuffs were also considered in our methods (Hörtenhuber et al. 2011). These farm models are also the basis for the evaluation of water demand. The calculation of the water demand is based on studies from Hoekstra et al. (2011) considering water use in different contexts. First, we considered "blue" water, which is consumed during production processes, including irrigation, cooling and water consumed in cleaning. Secondly, we included "green water" from precipitation and evapotranspiration of different crops and soil types (Asamer et al. 2011). Thirdly, we accounted for a theoretical water fraction called "grey" water that is necessary to dilute or assimilate a certain amount of pollutants in water bodies (see Mekonnen and Hoekstra 2011; Hoekstra et al. 2011, Chenoweth et al. 2013; Hörtenhuber et al. 2014). The virtual water demand is the sum of blue, green and grey water in liter per meal portion (500 kcal), whereas the climate effects defined by the GWP is depicted in kg CO_2eq per meal portion. The baseline for the identification of promising measures that entail a reduction of GHGE and water consumption is the conventional production system for beef and pork meat, as well as vegetables and potatoes. Natural gas for cooking and the average German electricity mix were considered as the baseline in both variants. An incremental plan was established that considered the use of organic livestock-based and vegetarian ingredients and the implementation of renewable energy sources, as well the use of frozen products.

3. Results

Following our transdisciplinary approach, we started with the identification and discussion of the most important questions to be addressed in the LCA. Intensive discussions resulted in the development of an incremental action plan that offers incentives to food service companies in order to implement feasible and ecologically responsible choices based on a baseline. We focused on three key questions:

1. What are the potential net savings from the implementation of green electricity in the canteen?
2. How much is the environmental impact of organic products especially organically produced meat?
3. What is the GHG mitigation potential of regionally produced vegetables versus imported ones?

The results for the GWP and the water demands of the vegetarian conventional (VC), vegetarian organic (VO) and non-vegetarian with conventional ingredients (MC) and the non-vegetarian plate with organic ingredients (MO) are presented in Figure 2. The column on the very left of each variant represents the initial baseline from which the incremental reduction measures of each indicator started. The baseline maximum magnitude is reached by the meat meal resulting in a water demand of 2,000 l and in a GWP of about 1.60 kg CO_2eq per portion. The baseline maximum of the vegetarian plate portion is considerably lower with a demand of 600 l water and 0.7 kg CO_2eq per portion. Farming stages contribute significantly to both indicators, especially for the meat variant.

The results show that the total sum of blue, green and grey water demand of the baseline meat plate is more than 70 % higher than the water consumption of the baseline vegetarian plate resulting in a potential water reduction of approximately 1,200 l per plate portion The virtual water demands are dominated by the water consumption of the agricultural production stages, whereas processing energy and transport have no visible impact on the overall water use for the vegetarian and the non-vegetarian plates. Although couscous contributes to only 14 % to the total energy content of the meal, it is related to a water consumption that is 300 l per portion. A switch to organic meat leads to a reduction of the virtual water demand of 305 l (17 %). Less significant would be a switch to organically produced vegetables where a reduction of 8 % or 50 l per portion could be obtained. According to our method, the water demands of the specific ingredients show wide ranges, the highest being for beef. In general, the overall virtual water demand and the advantages related to organic production systems were found to be related to grey water. Further, the results for livestock products are mainly dependent on the composition and quantity of the animal's diets and the related grey water content of the feedstuffs.

Our methodological approach to climate impacts (in terms of GWP) offered a more detailed picture in relation to potential ecological reduction measures based on the plate portion. The carbon footprint of the baseline is represented on the very left of the right half of Figure 2, which is followed by three respective potential reduction measures. Apart from beef and pork dominating on total GHGE per portion, cooking energy has a significant effect on total emissions per portion. Here, cooking energy is the sum of all preparation processes in the canteen kitchen including chilling and regeneration techniques, which are used in our project partner's food service and catering company. The implementation of organic meat was found to have a GWP reduction potential of more than 20 % (0.353 kg CO_2eq) including slightly lower cooking losses of the organic meat. The results show, that potatoes contribute 1-2 % of the total GHGE in the meat variant, which indicates negligible opportunities for reduction. The implementation of green electricity for the cook&chill and re-heating techniques has a potential reduction of 7 % related to the baseline, however in combination with organic meat and potatoes the reduction is about 40 % compared to the baseline. A total switch in the regenerative energy source in the canteen combined with organically produced ingredients reduces total GHGE by slightly more than 1.0 kg CO_2eq which is still higher than the vegetarian baseline.

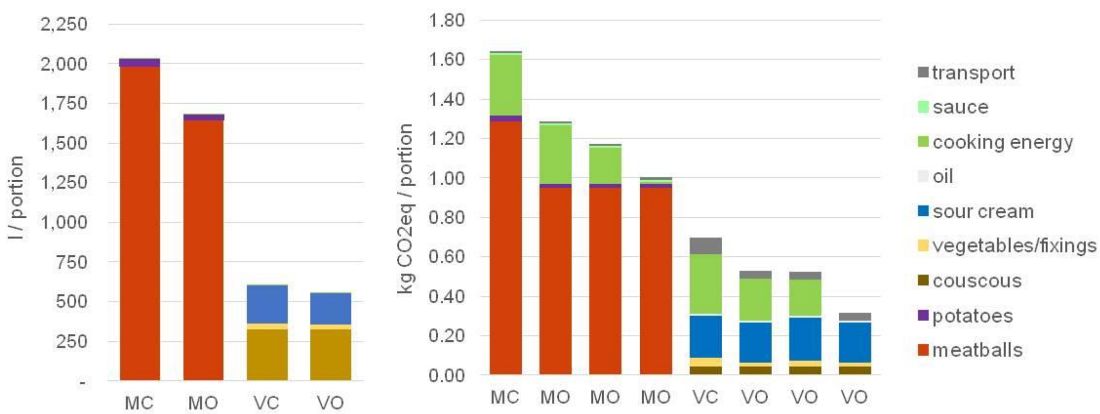

Figure 2. Virtual water demand (left part of the figure) and Global Warming Potential of vegetarian and non-vegetarian meals per portion after cooking, chilling and re-heating. Columns represent different plate variants and show incremental reduction measures. Conventional and organic ingredients were compared. MC: meat meal conventional; MO: meat meal organic; VC: vegetarian meal conventional; VO: vegetarian meal organic.

In the vegetarian meal variant, sour cream is of animal origin and contributes 0.3 kg CO_2eq to the total GWP. It can be seen from Figure 2, that a switch to sour cream from organic milk has a negligible reduction potential overall. The analysis shows that transport makes a higher contribution to the total GHGE per portion in vegetarian conventional and organic plates than in the non-vegetarian plates. Nevertheless, emission reductions due to the regional production of ingredients (presented in the first column) are negligible in comparison to the baseline and can easily be outweighed by calculation and data uncertainties. Although frozen vegetables from organic production show higher GHGE compared to the fresh organic variant, GHGE are found to be lower than a fresh conventional vegetables mix imported from Spain (regarding GHGE from agriculture and related upstream processes). However, in the case of a typical vegetarian meal with couscous and sour cream, no GWP reduction effect of using either fresh or frozen vegetables could be shown in our analysis. A considerable reduction potential is employing renewable energy to prepare vegetarian foodstuffs in the canteen kitchen. A switch from the average German power mix to green electricity for cooking, chilling and re-heating techniques in vegetarian plates could reduce the GWP of the vegetarian plate by 60 %.

4. Discussion and conclusion

Our current food systems are complex and ecologically, economically and socially unsustainable. Today, there are more people suffering from overnutrition and obesity than from undernutrition and related diseases (Searchinger et al. 2013). Due to the rising trend in out-of-home eating, food service companies are important stakeholders promoting sustainably produced and healthy foods.

Our results show that in addition to general suggestions of environmentally friendly measures, specifications on the energy mix used in the canteen kitchen should be considered. Substantial reductions of the climate effect, at least 22 %, can be achieved by switching from non-vegetarian to vegetarian plates (e.g. Stehfest et al. 2009, Carlsson-Kanyama and González 2009). Virtual water demands can be reduced by 17 % by shifting meat-eating habits from quantity to quality, specifically high quality organic meat., Beyond the impact of meal ingredients (directly related to consumer preferences), reduction potential also exists in the preparation of dishes and the selection of raw materials (an executive chief's decision). In German kitchens green electricity can reduce GWP considerably, because energy consumption from food preparation is a major contributor to total GWP, with a maximum of 65 % in vegetarian dishes and up to 30 % for non-vegetarian dishes. The virtual water demand is the lowest for the organic vegetarian dish with net gains of 1.126 – 1.431 l water per portion compared to the non-vegetarian variant. Plastically speaking, these amounts of water could be used to fill at least six bath tubes. The results show, that the virtual water demand of meat and vegetal ingredients is dominated by the amount of the grey water, which is calculated as the theoretical assimilation of nutrients in water bodies. Represented by grey water demand, the assimilation of nitrate and other harmful substances (such as pesticides) is 52 % and 33 % for conventional and organic meat ingredients respectively. For conventionally grown cereals and vegetables the share of grey water is 63 % and 41 % for organic products. It is evident that there is a close relation between our virtual water demand methods and the eutrophication potential commonly calculated in life cycle assessments (e.g. Saarinen et al. 2012).

One methodological limitation of our approach is that we assessed the environmental performance and reduction measures of lunch plates based on an isocaloric comparison instead of using a unit based on the nutritional quality of vegetarian and non-vegetarian plates. A detailed analysis of the environmental impacts based on the nutrient content of foods is desirable for further research. Nevertheless, carbon footprint and virtual water demands show similar performances regarding the incremental reduction plan based on the caloric value of meal portions. A relevant difference is that in order to reduce the virtual water demand per meal plate one would not consider the implementation of green. In a sensitivity analysis we considered the use of water power solely for the cooking, chilling and regeneration processes based on Mekonnen and Hoekstra (2012). The results showed that there is no meaningful impact on the total water demand for the non-vegetarian or vegetarian meal due to the use of hydropower.

One of the research questions addressed the differences between fresh and frozen vegetables, as this measure is easily implemented and reasonable in terms of cost from a food service company's perspective. We further wanted to evaluate the potential of frozen organic vegetables for consumption of organic vegetables produced regional but consumed outside the growing season. Here, we relied mainly on primary data and uncertainties were considered high. Although we found that frozen organic vegetables had a lower climate impact than conventional fresh vegetables and a slightly higher GWP than organic fresh vegetables, we did not consider the implementation of frozen vegetables as a major measure to reduce greenhouse gas emissions of food services companies based on a comparison of plate portions. We did not consider a detailed assessment of refrigerants that are related to a high amount of GHGE and which might lead to even higher GHGE from frozen products. The results show that the import of fresh organic non-seasonal vegetables including long distance transport would only have a small reduction potential compared to imported conventional vegetables. The latter would be overshadowed by uncertainties in the preparation process and waste along the supply chain and the use of animal-based side dishes such as sour cream.

This transdisciplinary approach enabled us to (re)act closely to the food service demands related to sustainability indicators. According to the manager of the food service company, data on the environmental impact is needed to directly inform the consumers in the canteen or to integrate labeling schemes directly in the menu. According to experiences of the food service company, consumers felt good about the fact that vegetables in the non-vegetarian meal are from organic production. An additional use of organic meat would require a much higher price per lunch portion and so far only few costumers are willing to pay more in our example. However, our results show that for the carbon footprint and virtual water demands, implementing organic potatoes has no potential reduction effect. Consumers tend to underestimate and misinterpret the complex interactions between raw material production, imported and regionally produced vegetables, and food processing. Integration of consumers in a transdisciplinary working group could help overcome this difficulty and could help researchers to better understand the preconditions under which consumers are willing to pay a higher price for sustainably produced food in canteens.

We conclude that taking environmentally responsible choices and measure for a reduction of the carbon footprint and virtual water demands in food service companies rely strongly on regional infrastructural constraints and company’s internal preferences and strategies. The outcomes of our calculations and the valuable discussions with the partners provided the basis for the establishment of a 5-minute-check online tool where small and medium sized food service companies are able to evaluate the performance of their supply chains in terms of GWP (http://klimae3.fiblprojekt.de/).

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Optimal practicable environmental model for canned tuna products

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ABSTRACT
This study aims to present an incorporated model between multi-objective linear programming (MOLP) and life cycle assessment (LCA). The model is to determine an appropriate choice of variables involved in the packaging system for canned tuna products. By the demonstration through a single-serve two-piece can, the results showed that the MOLP technique yielded feasible solutions in term of quantities of raw materials required to use in two-piece can for tuna products with satisfaction of various constraints pertinent to the production. The optimal solutions relating to objective functions of the environmental impact's carbon footprint and total economic cost relating to the packaging production are 4,767 kg CO_2 eq and $US 8,994.57[1] respectively. To this end, such optimal practicable environmental model can be beneficial as a decision-support tool not only to the production of canned tuna packaging, but also to other businesses alike to achieve an optimal packaging solution that compromises both economic and environmental aspects.

Keywords: multi-objective linear programming (MOLP), carbon footprint, economic cost, two-piece can, optimal solution

1. Introduction

For many manufacturers over the global regions, the environmental impact throughout a packaged-product's life cycle has become as important issue due to the present packaging directives and regulation for the environmental impact of all products. The manufacturers are not only experiencing an increasing concern in order to alleviate the environment pressure, but they have to seek ways to survive in the market competitiveness by reducing the production cost and achieving the high profit. Upon reacting to such conflict of interest, it is necessary to the manufacturers to make a decision in such a way that the environmental and economic issues are compromised.

Carbon footprint is now being applied in many products and has become part of public consciousness. It is an acclaimed measure of the emitted greenhouse gases (GHGs) produced through burning of fossil fuels for electricity, heating and transformation, and so on. The extent of GHG is expressed as kilograms of carbon dioxide equivalent (kgCO_2 eq) (Muthu et al., 2011). The approach for calculating the product carbon footprint into four steps: process mapping, determination of the problem boundaries, data collection, and calculation for carbon footprint. Carbon footprint is therefore the environmental information delivered from the product manufacturers to their consumers.

Costing issue is one of main considerations for most product manufacturers. In general, a higher product production cost may be incurred if manufacturers determine to opt for a superior packaging system in functionality in order to maintain the product quality. Also, it usually occurs that such packaging selection based upon the innovation concern can result in a contradiction as the environmental implication is increased. For instance, excessive product protection than necessary (over-packaging) is chosen and the packaging waste management at the end-of-life can be difficult. It is therefore important for manufacturers to select the right packaging system that can increase consumer satisfaction while having minimized environmental impact and economic cost (Monte et al., 2005; Poovarodom et al., 2011).

A single-served canned tuna has been responded the modern life-style. Through the product's life span, the associated costs related to the production of the canned tuna packaging include costs of raw materials, labors, manufacturing and operation, and last but not least, transportation. In the environmental aspect, all said activities of the canned tuna production could induce GHG emissions that directly influence the environment condition. It therefore has driven pressures to the food industry in order to reduce the carbon footprint associate with the product (Poovarodom et al. 2011).

Many studies for the environmental impact in food industry have emphasized solely one aspect without taking others into account. Poovarodom et al. (2011) showed that the manufacturing process of retort pouches and cups produced 60% and 70% less greenhouse gas emissions than that of metal cans. However, the overall carbon footprint of canned tuna in retort cups was 10% and 22% less than that in metal cans and retort pouches, respec-

[1] $US 8,994.57 (Equivalent to 287,826.13 THB; $US 1 = 32 THB)

tively. As the result, the retort cup packaging system possessed a significant advantage over metal can and retort pouch in term of overall GHG emissions. Packaging and its associated processing constituted significant fractions of the product's carbon footprint, ranging from 20% to 40%.

Zabaniotou and Kassidi (2003) compared the environmental impact of two egg packages made of polystyrene and recycled paper. The study indicated that recycled paper eggcups had less environmental impact than polystyrene ones. This is because, throughout its life cycle, the polystyrene eggcup had more contributions to acidification potential, and winter and summer smog, while recycled paper eggcup had contributions to heavy metal and carcinogenic substances.

Monte et al. (2005) adopted the life cycle assessment to determine a choice of coffee packaging. By comparing coffee packaging in five categories: 125-g cans, 250-g cans, 3-kg cans, cans with 36 single-use coffee servings, and poly-laminate bags with 40 single-use coffee servings (280g). The study indicated that the bigger packaging size of the metal cans, the more reduction of environmental impact. With a bigger size is only available for niche market and a smaller package is a marketing prerequisite, a laminated plastic bag is therefore recommended as an alternative, due to slight increase for the environmental impact.

Linear programming (LP) model can be incorporated with the life cycle assessment (LCA). This is because LCA is based on linear relationships between activities and environment burdens. LP can be used to allocate environmental impacts in the impact assessment of LCA. The LP solution not only gives the environmental optimum of system as a part of the improvement stage, but also incorporates economic and social aspects of the system (Azapagic and Clift 1988; Azapagic and Clift 1995). However, goals toward the economic and the environmental aspects usually contradict to each other. Multi-objective linear programming (MOLP) can be employed to resolve such conflicts and to determine the best compromised solution. MOLP is a multiple criteria decision making. It is concerned with mathematical optimization problems involving minimizing or maximizing multiple objective functions simultaneously. Optimal decisions need to be taken in the presence of trade-offs between two or more conflicting objectives. This approach provides a decision making tool which can help the businesses to identify a path to sustainable development by establishing good trade-off between economic cost and environmental performance in term of carbon footprint.

In this study, MOLP is developed as an optimal practicable environmental model in order to determine the appropriate choice of packaging system by using two-pieces can for the canned tuna products as a case study. The trade-off between two objectives between minimized total cost and minimized carbon footprint is considered. The aim of the study is to enhance the application of MOLP at the early stage of a new product development and a product improvement.

2. Methods

2.1. Functional unit and system boundary

The demonstration through a single-serve two-piece can packaging is used. The packaging features consist of chrome-coated steel for can and a pull-ring aluminum tab closure. The two-piece can considered in this study is the plain can (no printing on it). The 85-gram packaging size is considered. Such size is regularly available for retail and becomes standardized for one single serving (Figure 1).

The functional unit is defined as 90,000 cans, equivalent to one pallet of end shell and three pallets of can body. The model is developed under assumptions in which the production of tuna, filling, storage, and disposal are excluded. The system boundary is considered as cradle-to-gate (Figure 2). It spans the acquisition of raw materials, manufacturing, and transportation. Other additional packaging types used in each individual process such as tier sheets, low-density polyethylene (LDPE) film, paper pallet tags, and polypropylene (PP) strapping are also included in the study.

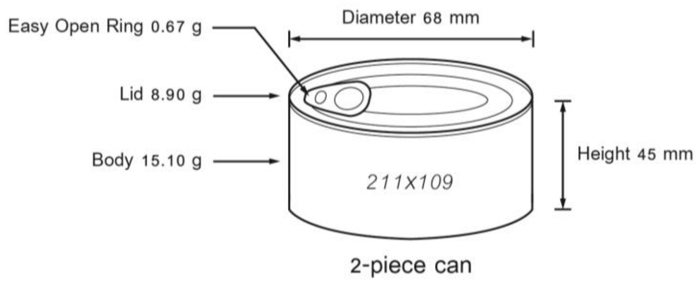

Figure 1. Two-piece can that use in the study.

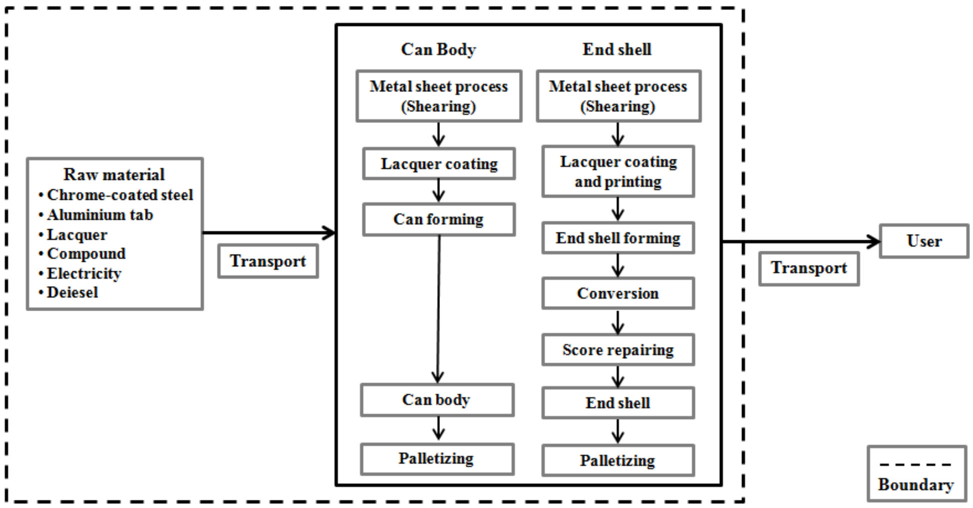

Figure 2. System boundary of two-piece can for tuna products.

2.2. Inventory analysis

Life cycle inventory (LCI) is collected from primary data (Table 1.). It is derived from questionnaires and by mean of interviewing personnel from can suppliers. Secondary data from literature and database are also used. Emission factors are obtained from Thai National Database (Thailand Greenhouse Gas Management Organization 2013) and some are provided from can suppliers.

Economic costs considered in this study include costs of materials, energy consumption, operation, and transportation. The unit costs are collected from can manufacturers.

Table 1. Information on raw materials and two-piece cans

Raw material	Source	Type of shipment	Distance (km)
TFS steel sheet	Thailand	Truck (10 wheels)	204
Lacquer A for can body	Thailand	Truck (6 wheels)	20
Lacquer B for can body	Thailand	Truck (6 wheels)	69
Lacquer C for end shell	Thailand	Truck (6 wheels)	36
Lacquer D for end shell	Thailand	Truck (6 wheels)	90
Lacquer E for end shell	Thailand	Truck (6 wheels)	36
Aluminum Tab	America	Ocean freighter	16,469
Solvent	Thailand	Truck (6 wheels)	30
Wax	Thailand	Truck (4 wheels)	50
Tab lube	America	Ocean freighter	16,718
Compound	Thailand	Truck (6 wheels)	36
Ink	Thailand	Truck (6 wheels)	80
Layer paper sheets	Thailand	Truck (10 wheels)	140
Linear low-density polyethylene	Thailand	Truck (6 wheels)	18
Polyethylene	Thailand	Truck (6 wheels)	33
Plastic strapping	Thailand	Truck (4 wheels)	30
Paper bag	Thailand	Truck (6 wheels)	20
Wood pallet	Thailand	Truck (6 wheels)	7

2.3. MOLP model

As previously mentioned, two objective functions considered in the study are total carbon footprint and total economic cost. The MOLP model is able to formulate and yield the optimal solution by using spreadsheet solver in Microsoft Excel. Steps for developing the model are described in the following.

Decision variables: The decision variables are quantity of raw materials and packages. The decision variables are expressed as X_j (quantity of raw materials in production of can body type *j* in kg) and Y_j (quantity of raw materials in production of end shell type j in kg). The decision variables are showed in Table 2.

Table 2. Decision variables for two-piece can system.

Production	**Decision variable**	**Description**	**Unit**
Production of can body (X_j)	X_1	Quantity of TSF steel sheets	kg
	X_2	Quantity of Lacquer A	kg
	X_3	Quantity of Lacquer B	kg
	X_4	Quantity of solvent	kg
	X_5	Quantity of wooden pallets	kg
	X_6	Quantity of layer paper sheets	kg
	X_7	Quantity of top wooden frames	kg
	X_8	Quantity of stretch film	kg
	X_9	Quantity of plastic strapping	kg
	X_{10}	Quantity of paper pallet tags	kg
Production of end shell (Y_j)	Y_1	Quantity of TSF steel sheets	kg
	Y_2	Quantity of lacquer C	kg
	Y_3	Quantity of lacquer D	kg
	Y_4	Quantity of lacquer E	kg
	Y_5	Quantity of solvent	kg
	Y_6	Quantity of ink	kg
	Y_7	Quantity of compound	kg
	Y_8	Quantity of Al Tab stocks	kg
	Y_9	Quantity of tab lube	kg
	Y_{10}	Quantity of repairing lacquer	kg
	Y_{11}	Quantity of wooden pallets	kg
	Y_{12}	Quantity of wrapping paper	kg
	Y_{13}	Quantity of shrink bag	kg
	Y_{14}	Quantity of stretch film	kg
	Y_{15}	Quantity of plastic strapping	kg
	Y_{16}	Quantity of paper pallet tags	kg

Objective functions: In order to determine the optimal solution for two-piece can system, two objective functions are minimized total carbon footprint (Z_1) and minimum total economic cost (Z_2).

$$Min\ z_1 = \sum_{j=1}^{n} e_{ij} x_j \qquad \text{Eq. 1}$$

$$Min\ z_2 = \sum_{j=1}^{n} c_{ij} x_j \qquad \text{Eq. 2}$$

where e_{ij} is emission factors related to raw materials, electricity, and transportation of raw material type *j* with objective *i* (i.e., *i* = 1, 2); C_{ij} is unit costs of raw materials, transportation, and operation for two-piece can system.

Constraints: Constraints concern mass balance and assured quantity of raw materials for two-piece can.

$$\sum_{j=1}^{n} x_j = m_b \qquad \text{Eq. 3}$$

$$S_{j(\min)} \leq x_j \leq S_{j(\max)} \qquad \text{Eq. 4}$$

where m_b is process outputs; $S_{j(min)}$ and $S_{j(max)}$ are minimum and maximum quantity of raw material type *j* in each process.

Target values: Each objective has its target value (t_i). However, percentage deviation from each target value will be calculated. Later, the weighted percentage deviation of each objective function is also determined (Ragsdale CT 2007). In this study, the weights factors depend upon the importance of each objective and can be derived from the interview according to preference of can manufacturers. The weight of importance for total carbon footprint is 2.7 while the total economic cost has been given to 3.3.

$$\textit{Percentage Deviation} = \frac{\textit{Actual value} - \textit{Target value}}{\textit{Target value}} \qquad \text{Eq. 5}$$

$$\textit{Weighted percentage deviation} = \textit{Weight factor} \times \textit{Percentage deviation} \qquad \text{Eq. 6}$$

MINIMAX objective: It minimizes the worst-case values of a set of multivariate functions, possibly subject to linear constraints. The MINIMAX objective is going to minimize the maximum of weighted percentage deviation (Q) (Ragsdale CT 2007).

Objective function: *MIN: Q* Eq. 7

Constraint: *Weighted percentage deviation* $\leq Q$ Eq. 8

Decision variable: Q

The optimal solution: Considered by decision makers.

3. Results

In this section, the results of the study are divided into three parts. The first part showed target values of each objective. The second part indicated weighting scores. The third part showed the optimal solution that is compromised both objectives on the environmental and economic cost aspects for two-piece can.

With LP technique by considering a single objective function and solving to find the solution of each objective, The results can be found as shown in Table 3. It indicates that if only total carbon footprint as a single objective function is determined, the model yielded the value of 4,745 kg CO_2 eq while the total economic cost was $US 8,994.40 (Equivalent to 287,820.65 THB; $US 1 = 32 THB). However, if the total economic cost as a single objective function is determined, the model yielded the value of $US 8,993.31 while the total carbon footprint was increased to 4,781 kg CO_2 eq. According to the results, it seems apparent that both objectives contradict to each other. As the result, the LP model could not yield the optimal solution for both objective functions simultaneously. If there is one objective function producing a better result, the solution of the other objectives becomes worse. With this reason, MOLP is taken in in order to solve for the optimal solution. However, the weighting factor of importance is needed to apply as earlier mentioned in Section 2.3.

The weighting scoring is a valuable decision-making tool. It is used to evaluate alternatives based on specific evaluation criteria weighted by importance or priority (Zimmer DA 2011). By evaluating alternatives based on performance with respect to individual criteria, a value for the alternative can be identified. The weighting value enables an organization to narrow the list of options using criteria such as cost, quality and efficiency. In this study, the weighting scores of both objective functions are provided by can manufacturers. The weights of the total economic cost and total carbon footprint are 3.3 and 2.7, respectively. After applying the trade-off between two objective functions for two-piece can system, the optimal solution of MOLP model functions is shifted as shown in Table 4.

Table 3. Target values of two objective functions.

Objective function	Total carbon footprint (kg CO_2 eq)	Total economic cost ($US)	Target values	Unit
Minimized total carbon footprint	4,745	8,994.40	t_1 = 4,745	kg CO_2 eq
Minimized total economic cost	4,781	8,993.31	t_2 = 8,993.31	$US

Table 4. Objective values (Z) and the maximum weighted percentage deviation (Q)

Objective function	Weighting	Optimal objective value (Z)	Unit	Maximum weighted percentage deviation from the target values (Q)
Minimized total carbon footprint	2.7	4,767	kg CO_2 eq	0.000494547
Minimized total economic cost	3.3	8,994.57	$US	

The MOLP model represented the optimal solutions for two-piece can system. By considering the two objectives, with the functional unit of 90,000 cans, the subsequent results were 4,767 kg CO_2 eq for total carbon footprint (Z_1) and $US 8,994.57 for total economic cost (Z_2). The maximum weighted percentage deviation from the target values (Q) was 0.000494547 or alternatively speaking that the solutions were within approximately 0.000494547% of achieving the target solutions for the both objective functions. These solutions of this study were depending on weighting score determined by can manufacturers. Thus, it enables a decision maker to explore several solutions by adjusting the weighting scores. According to the optimal values of decision variables in the case study can be determined as showed in Table 5. The results indicate that all quantities of decision variables were within the assured range for the production. It is noted that for the production of can body (X_i), the quantity of TSF steel sheets was decreased. On the other hand, the quantities of lacquer A, lacquer B and solvent in this process were increased. For the production of end shell (Y_i), the quantity of TSF steel sheets was decreased. In addition, the quantities of Lacquer C, Lacquer D, Lacquer E, ink and compound in this process were also increased.

Table 5. Optimal quantities of decision variables for two-piece can system (functional unit = 90,000).

Production	Decision variable	Description	Optimal value	Unit
Can body (X_j)	X_1	TSF steel sheets	1038.85	kg
	X_2	Lacquer A	20.34	kg
	X_3	Lacquer B	13.65	kg
	X_4	Solvent	0.92	kg
	X_5	Wooden pallets	180.09	kg
	X_6	Layer paper sheets	66.56	kg
	X_7	Top wooden frames	36.02	kg
	X_8	Stretch film	2.16	kg
	X_9	Plastic strapping	0.36	kg
	X_{10}	Paper pallet tags	0.04	kg
End shell (Y_j)	Y_1	TSF steel sheets	677.58	kg
	Y_2	Lacquer C	8.25	kg
	Y_3	Lacquer D	7.13	kg
	Y_4	Lacquer E	11.35	kg
	Y_5	Solvent	0.79	kg
	Y_6	Ink	0.50	kg
	Y_7	Compound	2.40	kg
	Y_8	Al Tab stocks	56.59	kg
	Y_9	Tab lube	0.75	kg
	Y_{10}	Repairing lacquer	1.86	kg
	Y_{11}	Wooden pallets	22.31	kg
	Y_{12}	Wrapping paper	0.62	kg
	Y_{13}	Shrink bag	3.69	kg
	Y_{14}	Stretch film	6.69	kg
	Y_{15}	Plastic strapping	1.12	kg
	Y_{16}	Paper pallet tags	0.13	kg

4. Discussion

The optimal solution of the developed MOLP model with two objective functions of total carbon footprint and total economic cost for 2-piece can system is evaluated. The results show that the optimal value on objective functions has improved. However, further improvement can be established if the constraints are not too much restricted. The region of feasible solution will be subsequently expanded. In addition, a change in the weighting factors of importance may also be implemented in order to obtain a better solution.

5. Conclusion

The MOLP can be used as the optimal practicable environmental model for compromising relationship between the two main objectives involving the total economic cost and the environmental performance in term of total carbon footprint. The developed model in the study represented the optimal solutions for two-piece can system for the canned tuna products. By considering two objectives function in term of functional unit of 90,000 units, the results showed that the optimal solutions were 4,767 kg CO_2 eq for the total carbon footprint and $US 8,994.57 for the total economic cost. With the maximum weighted percentage deviation from the target values of 0.000494547%, the MOLP model yielded the optimal solution in term of quantities of raw materials required to use in the production of the two-piece can with satisfaction of various constraints pertinent to the production.

6. Acknowledgement

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Specialty Food Ingredients – Environmental Impacts and Opportunities

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ABSTRACT

Specialty food ingredients (SFIs or just 'specialty ingredients') such as emulsifiers, food enzymes, hydrocolloids and cultures are typically applied in small amounts (<1% w/w of the final food product). Generally, little attention has been given to specialty ingredients in the LCA community (Foster et al 2006, BCFN 2011). DuPont Nutrition & Health (N&H) is a leading producer of specialty food ingredients worldwide. During the last five years, DuPont N&H has completed cradle-to-gate LCAs of a wide range of specialty ingredients, based on both attributional and consequential modeling. The average carbon footprint of DuPont N&H products is 3-4 kg CO_2e per kg (cradle-to-gate) but they only represent a small share of the final consumer products carbon footprint. More importantly, most solutions enable significant reductions of our customer's footprint by enabling replacement of animal derived raw materials (with e.g. soy protein), increasing processing efficiency or enabling reduced food waste in retail and households. DuPont N&H has identified more than 70 'sustainable solutions' and LCA screenings of 40 cases show that they can help customers reducing between 10 and 100 kg CO_2e per kg specialty ingredient applied.

Keywords: Specialty Food Ingredients, Life Cycle Assessment, Carbon Footprint, Food Waste, Sustainable Food

1. Introduction

A significant number of food companies apply life cycle assessment (LCA) to quantify the environmental burdens associated with their products. LCA studies have been published on a wide range of food, including dairy products, meat products, vegetables, fruits, bread, alcoholic and non-alcoholic beverages, confectionary etc. (Foster et al 2006, BCFN 2011). Few studies, however, are available on specialty ingredients such as:

- Emulsifiers (e.g. used to strengthen and soften the dough in bread)
- Food enzymes (e.g. bakery enzymes used to extend periods for which breads stay fresh)[1]
- Hydrocolloids (e.g. pectin used as gelling agent in jam and marmalade)
- Bacteria cultures (e.g. acidifying cultures applied in the production of yoghurt and fresh fermented milks)
- Antimicrobials and antioxidants (e.g. rosemary extract used to prevent pathogens and oxidation in meat)
- Reduced calorie sweeteners (e.g. xylitol used as sweetener in chewing gum).
- Soy protein (e.g. soy protein isolate used to replace animal protein in different food applications)

Apart from a few examples such as soy proteins and some sweeteners, specialty ingredients are typically used in small amounts (<1% w/w) to obtain certain functions such as extended shelf-life/freshness or improved taste, texture and mouth feel. In most existing LCAs of food products, impacts associated with SFIs are only modeled coarsely. But it is also important for food producers to understand the environmental impacts associated with SFIs, and more importantly to be aware of the opportunities they bring in terms of replacement of high impact raw materials, improvements in processing efficiency and not least reduction of food waste.

As a leading producer of SFIs, DuPont N&H decided three years ago to carry out cradle-to-gate LCAs on all main product categories. This paper shows examples of results from these studies as well as results from LCA screenings of the use stage. In the discussion part of the paper, attention will be given to methodological challenges.

[1] Several scientific LCA studies on enzymes including food enzymes are available in the literature (Jegannathan and Nielsen 2013).

2. Methods

In terms of cradle-to-gate LCA studies there has been completed six studies which have been third party reviewed in accordance with the ISO 14044 standard (two additional studies are in review). Our goal is that all our main product categories should be analyzed through 3rd party reviewed ISO 14044 compliant LCA studies by 2015. In addition, 40 LCA screenings of use stage impacts have been completed for a range of products and applications that we call sustainable solutions. The following sections address key methodological choices related to our cradle-to-gate LCAs as well as the LCA screenings of the use stage.

2.1. Goal and Scope

Our cradle to gate (or cradle-to-customer) LCAs include all life cycle stages from raw material acquisition to final dispatch of products to our customers and includes in many cases even outbound transport to the average customer. The functional unit can vary from study to study but reflects a reference flow of 1 kg specialty ingredient in all studies. Specialty ingredients are sold because of their function which is the reason why we often describe them as ‘solutions’ rather than ‘products’. The functionality is therefore always described in detail, and relevant adjustments in the reference flow are made when comparisons are made between different SFIs.

2.2. Life Cycle Inventory (LCI)

In DuPont N&H consequential and attributional life cycle inventory modeling are seen as complementary rather than competing which will be explained later. Both approaches are therefore included in our detailed third party reviewed LCA studies as well as LCA screenings of the use stage.

In consequential modeling co-product allocation is consistently avoided by system expansion (substitution) and constrained processes are excluded from the analyzed product system. As an example, this means that hydropower would not be included in the grid mix for electricity in a country where all hydropower is fully utilized (e.g. Norway). Modeling of indirect land use change (ILUC) is only included in sensitivity scenarios.

Attributional modeling represents an approach where co-product allocation is typically handled with allocation, and where no distinctions are made between constrained and unconstrained processes (Sonnemann and Vigon 2011 p74). In DuPont N&H economical allocation is generally used for handling of co-product allocation in our attributional models, which is consistent with the ecoinvent V3 default allocation model (Weidema et al 2013).

Background processes have generally been modeled based on data from the ecoinvent V2.2 database (in all studies completed before 2014), but other data have been used when these have been deemed to represent a higher level of data quality. This includes, for example, data on country-specific marginal and average electricity which has been modeled separately in collaboration with 2.-0 LCA consultants (Schmidt et al. 2011). Ongoing studies are based on Ecoinvent 3 which allows for more consistent modeling, where either economical allocation or system expansion is used in all background processes.

2.3. Life Cycle Impact Assessment (LCIA)

The Recipe midpoint model has been used as the default method for impact assessment (in all studies completed before 2014) but other LCIA methods, including end-point methods, have been applied for sensitivity studies. Ongoing LCAs applies the ILCD LCIA method available in SimaPro 8 which is applied as it is recommended by the European Product Environmental Footprint (PEF) Guide and is a result of a consensus project aimed at establishing current best practice for LCIA (European Commission 2012).

2.4. Use stage modeling

In most cases, one type of specialty ingredient can typically be used in many different food products where it can have various functions. Furthermore, specialty ingredients are often used in various combinations and in different parts of the world. Needless to say, it is complex to model the use stage without focusing on concrete cases. The use stage has therefore been modeled separately and at screening level, focusing on specific applications in different parts of the world. Three scenarios are calculated for each solutions which represent attributional modeling, consequential modeling (incl ILUC) and consequential modeling (excl ILUC) which is chosen as our default model. The advantage of the different models is that it allows us to adapt to specific customer preferences. In terms of LCIA the screenings only address the carbon footprint based on characterization factors in the ILCD LCIA method in Simapro 8.

Conceptually, it is possible to distinguish three types of SFIs based on their functions: Specialty ingredients that mainly are applied to enable a substitution of raw materials (or modify recipes), ingredients that mainly aim at increasing processing efficiency and finally ingredients that serves to prolong the freshness or shelf-life of food products, see Figure 1.

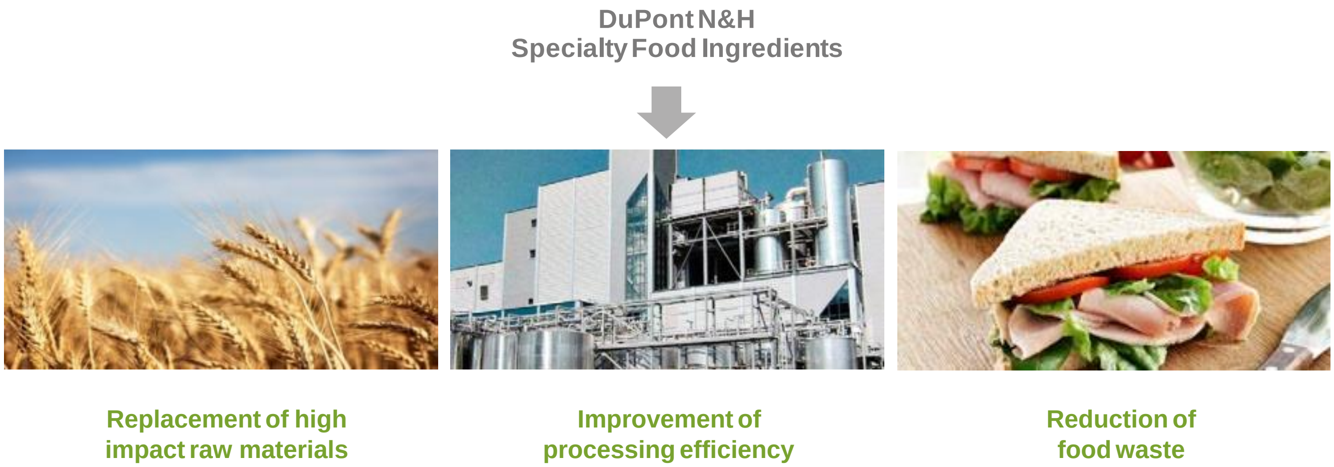

Figure 1. Conceptual model of interaction between SFIs (top center) and the food value chain with raw material providers (left), food producers (center) and finally end-consumers (right).

As illustrated the distinguishing feature between the three groups of SFIs depend on which part of the food value chain they mainly address:

- Upstream SFIs: Some SFIs address upstream impacts by enabling modifications in raw material composition e.g. by replacing animal protein and fat with vegetable protein and fat (Figure 1 left). This application can be divided in two sub-categories: direct upstream where the SFI itself constitutes the replacement (e.g. soy protein that replaces milk protein) and indirect upstream where the SFI has an indirect role allowing the replacement to take place (e.g. hydrocolloids that facilitate a reduction of animal fat or a replacement with vegetable fat in dairy, while maintaining similar taste and texture).
- Processing SFIs: Another group of SFIs addresses the food processing directly – e.g. by allowing for larger throughput and/or increased efficiency (Figure 1 center).
- Downstream SFIs: These are SFIs that address the food products’ freshness and their taste and texture as perceived by the end customers in a typical use scenario (Figure 1 right). This application has the potential to reduce food waste significantly, especially for categories of food products where the dominating reason for food waste is that it has passed a date label, has gone moldy or rotten, looked, smelt or tasted bad.

All three functions have a significant potential to promote sustainability in the food value chain – especially upstream and downstream SFIs that address the most important hot-spots of average food products (raw materials and food waste). A large generic group of SFIs such as food enzymes or hydrocolloids can have several of the above mentioned functionalities, while more specific products typically have a more narrow functionality related to one of the functions.

3. Results

3.1. Example of comparative LCA on Xylitol/Xivia®

One of the first LCAs completed by DuPont N&H was a comparative LCA on the sweetener xylitol, which was reviewed by a third party panel in 2011. Two production methods were compared – one based on a side stream from the pulp and paper industry (the wood based concept used by DuPont), and one based on corn cobs (used by most of our competitors). The study was completed by Earthshift consultants and applies attributional modeling but with the use of system expansion (substitution) for energy and waste related foreground processes. In this particular study, Impact 2002+ has been used for life cycle impact assessment (LCIA). A significant difference was anticipated, but it was a surprise to most that the wood based technology actually generated 85-99% lower impacts across all investigated impact categories (see Figure 2).

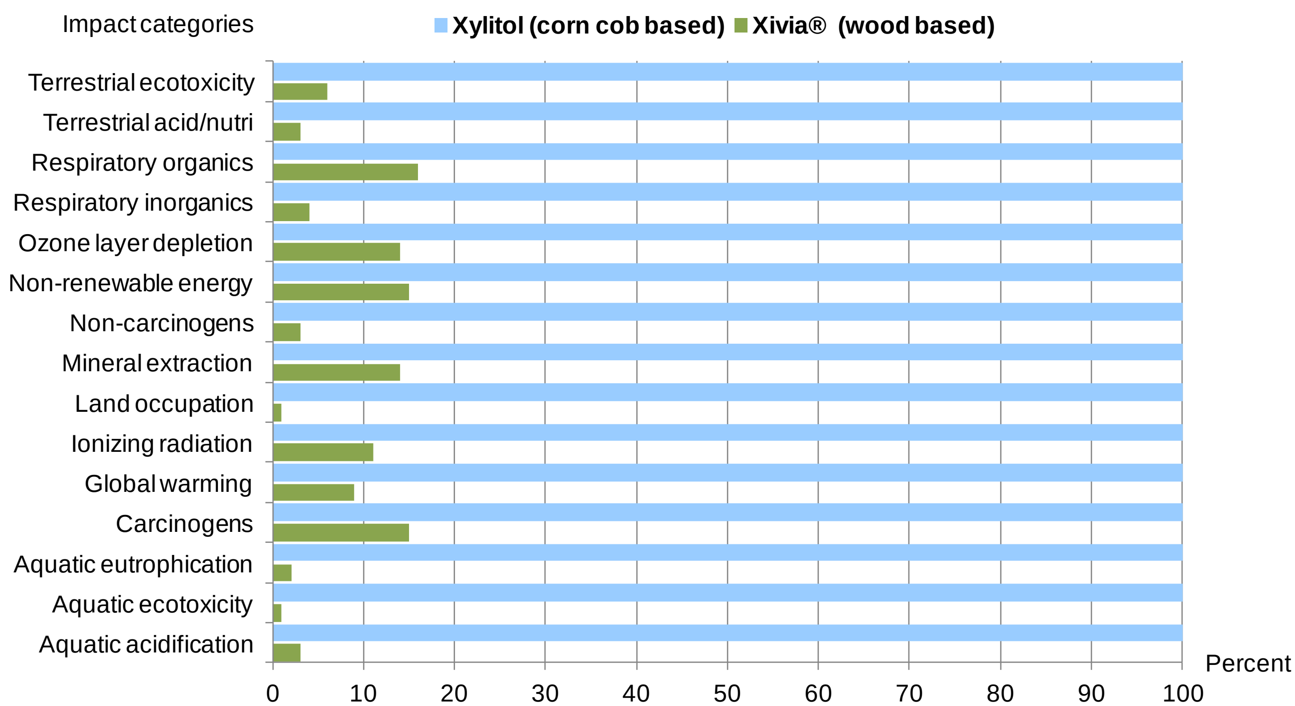

Figure 2. Characterized results from comparative LCA (based on Impact 2002+) of corn cob based xylitol and DuPont's wood based xylitol (Xivia®) based on Dahliwal, Hamilton and Laurin (2010).

In terms of global warming potential the wood based technology contributed 3.6 kg CO_2e per kg Xivia® (cradle-to-gate) while the corn cob based technology contributed 38.6 kg CO_2e per kg xylitol from cradle-to-gate (Dahliwal, Hamilton and Laurin 2010). Results also published in Danisco (2011) and Dahliwal and Thrane (2011).

3.2. Carbon footprint and hot-spots for different groups of SFIs

Based on findings from LCA studies of SFIs as well as DuPont's Corporate Footprint based on WRI & WBCSD (2004 and 2010), it can be concluded that average carbon footprint of DuPont N&H's specialty ingredients is in the range of 3 to 4 kg CO_2e per kg from cradle-to-gate, excluding ILUC. The main contribution comes from raw materials (53%), while processing represents 35% of the cradle-to-customer footprint. Inbound and outbound transport both represents 6% respectively (Thrane 2011, Dalgaard et al. 2013).

With a typical application level of less than 1% w/w of the customer's final product, this suggests that specialty ingredients in most cases have a modest contribution to a food product's total footprint. There are cases, however, where SFIs are used in larger amounts such as for soy proteins, and in the case of Xivia® in chewing gum applications.

Emulsifiers: For average emulsifiers, the raw materials (mainly vegetable oils) constitute about 80% of the cradle-to-gate footprint, and the average emulsifier has a carbon footprint of 3-5 kg CO2e per kg product from cradle-to-gate (Muñoz 2014). Both attributional and consequential modeling has been applied, but the results are within the range mentioned above in both LCI models.

Food enzymes: LCA of enzymes have been conducted for several years in DuPont Industrial Biosciences. In terms of carbon footprint, the most important life cycle stage for food enzymes are processing (fermentation, formulation and recovery) followed by raw materials use in the fermentation process. The carbon footprint ranges between 1 and 10 kg CO_2e per kg enzyme from cradle-to-gate mainly depending on the type and quantity of raw materials, the fermentation time and the concentration (Dettore 2014). Again both consequential and attributional modeling has been used and both approaches are represented within the indicated carbon footprint range.

Hydrocolloids: In terms of carbon footprint, the processing stage is important for most hydrocolloids. Processing aids also have a noteworthy contribution to the footprint, but it should be stressed that hydrocolloids are a very diverse group of ingredients with large variations in hot-spots and impacts. The typical carbon footprint ranges from 1-12 kg CO_2e per kg hydrocolloid from cradle-to-gate. This range covers many different hydrocolloids of which most have been analyzed through screening level LCAs apart from a third party reviewed LCA of pectin published in 2011 (Thrane 2011). Consequential modeling has been applied in combination with attribution modeling in the third party reviewed LCA which showed only a minor difference in impacts.

Cultures: Similar to most hydrocolloids, the processing stage represents an important contribution to the carbon footprint for cultures – mainly due to the energy use. Two types of direct inoculation cultures exist: Frozen cultures and freeze dried cultures, which are concentrated and therefore more efficient. Adjusted for functionality, the cradle-to-gate footprint for both types of cultures are in the range between 8 and 12 kg CO2e per kg. Cultures are added at rather small dosages in yoghurt and cheese; between 0.5 and 2 kg frozen cultures per 10,000 liters of milk. As for most SFIs their contribution to the final products footprint is therefore small. In a cradle-to-customer perspective, the carbon footprint for outbound transport can be important, especially for frozen cultures. The reason is that frozen cultures have to be stored at minus 50°C to 60°C and frequently require air transport – especially for transcontinental shipments. In cases where outbound transport involves airfreight, freeze dried cultures are likely to offer a low carbon footprint alternative to frozen cultures. Both consequential and attributional modeling has been applied and again no significant differences in impacts have been identified apart from what is covered by the suggested interval (Thrane 2012, Thrane 2013a, Thrane 2013b).

Reduced calorie sweeteners: For this category, only Xivia® has been analyzed and the study shows that the raw material- and processing stage represent the largest carbon footprint (due to energy use). The carbon footprint of Xivia® is within the same range as our average product, 3-4 kg CO2e per kg (Dahliwal, Hamilton and Laurin 2010). But as Xivia® is the only sweetener type that has been analyzed, it is not possible to conclude that this product is representative for the other products in this category.

Protein solutions: For soy protein concentrate (min 65% protein per dry matter) and soy protein isolate (min 90% protein per dry matter) the main contribution comes from the processing stage due to energy use. A detailed LCA is being completed but final results are not yet available. Preliminary results indicate that the carbon footprint is around 2-4 kg CO_2e per kg protein product. Consequential modeling suggests a lower impact compared to attributional modeling.

Apart from the product groups mentioned above, DuPont N&H also produces antimicrobials and antioxidants (covering products such as Natamax®, Nisaplin® and GUARDIAN® Rosemary Extract) as well as a range of other products that could be termed 'other' which include fibers, rare sugars, soy lecithin etc. Considering the large number of ingredients and applications, the scope has been limited to cover the above mentioned groups in the present article.

3.3. LCA screenings of avoided burdens (use stage impacts)

As mentioned, the use stage has also been modeled by LCA screenings focusing on the carbon footprint. So far, more than 70 solutions have been identified and 40 of these have been quantified by LCA screenings. In a typical case, 1 kg specialty ingredient contributes to avoiding 10-100 kg CO_2e (net) compared to a situation where the ingredient is not used. All LCA screenings have been made according to both attributional and consequential modeling, but the results presented in the following are only based on consequential modeling, which in most cases represent a more conservative estimate of potential savings.

Direct upstream SFIs: One example of direct raw material replacement is the use of Xivia® instead of corn cob based xylitol in chewing gum. The net avoided impact per kg applied Xivia® is the difference in impact between one kg corn cob based xylitol and one kg Xivia®, which amounts to 35 kg CO_2e. LCA screenings shows that the carbon footprint is reduced by 42-85% per kg chewing gum when switching from corn cob based xylitol to Xivia®. The large interval represents differences in xylitol content in the final recipe with all ingredients including gum base. The modeled xylitol content ranges from 5% to 55% (Dalgaard et al. 2013a). Another example could be the use of soy protein to replace milk or meat protein in different applications such as dairy products, sausages and burgers. Updated third party reviewed LCA results of soy protein isolate are not yet available, but our LCA screenings suggest that 1 kg isolated soy protein contributes to reducing 10 kg CO_2e (net) in application where it replace milk protein[2] and even more when replacing meat protein. An example of a beef burger, where the beef content is partly replaced with textured soy concentrate, shows that the carbon footprint of the beef can be reduced with as much as 35% (Dalgaard et al. 2013 a and b).

Indirect upstream SFIs: An example where the specialty ingredient has an indirect effect on raw materials could be a pectin based ingredient solution, such as GRINDSTED® SB555, that makes it possible to replace milk protein with vegetable protein in yoghurt, without compromising organoleptic properties. An LCA screening shows that 1 kg GRINDSTED® SB555 contributes to reducing 12 kg CO_2e per kg (net) in a specific application where it allows for using vegetable protein instead of milk protein in yoghurt. The solution allows the footprint of the final yoghurt to be reduced with 18%. Other solutions exist which enable replacement of meat protein in sausages, burgers etc. (Dalgaard et al. 2013b). Another example of indirect upstream substitution could be the enzyme Alphalase® AP4 applied in beer production. This solution makes it possible to brew a good quality beer where 60% of the malt is replaced with barley. LCA screenings show that 1 kg Alphalase® AP4 contributes to reducing 100 kg CO_2e (net) in this application and the footprint of the final beer is reduced by 12% (Dalgaard et al. 2013a).

Processing SFIs: An example of processing SFIs is FoodPro® Cleanline - an enzyme solution which can significantly reduce the impacts of ultra high temperature processing of milk in dairies (UHT). This solution prevents fouling in the UHT processing systems, leading to higher capacity while reducing the need for cleaning cycles that otherwise would require significant amounts of water, energy and cleaning chemicals. An LCA screening shows that 1 kg FoodPro® Cleanline contributes to reducing 35 kg CO_2e per kg (net) when applied in UHT processing. In a large scale dairy with 6 UHT lines, this translates in to annual net savings of 322 tons CO_2e (Dalgaard et al. 2013b).

Downstream SFIs: Finally an example of a solution that addresses downstream food waste is the use of emulsifier and enzyme blends in bakery products. Food waste is a major issue, and bakery products are one of the worst culprits for avoidable food waste. Approximately 30% of bakery goods like bread, cakes and cookies are wasted in the households and according to WRAP about 90% of this waste occurs because the products are not perceived as fresh (Gustavsson et al. 2011, Quested and Johnson 2009). DuPont has developed enzyme solutions that apart from providing the usual bread softness and crumb elasticity, also keep the bread moist longer. The enzymes are sold under the Powerfresh® brand name and are often used in combination with emulsifiers. An LCA screening has been completed for a use of 50% higher dosage of Powerfresh® (increase from 20 gram to 30 gram enzyme dosage per 100 kg flour) in toast bread in North America, enabling the bread to stay fresh for 21 days compared to 14 days with the lower dosage. Based on a model of the relationship between freshness and bread waste, calibrated with data from Quested (2013), it is estimated that the bread waste in retail and households is reduced by 30%. The LCA screening shows that the carbon footprint is reduced with 12% per kg bread consumed in this case – and several thousand kg CO_2e is avoided per kg extra enzyme that is applied. This example involves our newest and most sophisticated enzymes, but even standard enzymes (and emulsifiers) can make a huge difference in markets where enzymes are more rarely used, such as parts of Eastern Europe and Russia.

A new exiting product range with a significant potential for food waste reduction is protective cultures such as HOLDBAC® cultures used for spoilage and pathogen prevention in dairy and meat applications. LCA screen-

[2] In this example the milk protein is modeled based on the LCAfood database where the process "Milk powder, no quotas' has been modified and updated based on LCI data on milk from Schmidt and Dalgaard (2012) & Dalgaard and Schmidt (2012) using the consequential mode. If attributional modeling had been used instead the avoided burden would have been 15 kg CO2e per kg soy protein that replaces milk protein (Dalgaard et al. 2013 b).

ings of protective cultures applied to fresh cheese shows that the carbon footprint can be reduced with nearly 5% per kg consumed cheese – and 220 kg CO_2e is avoided per kg HOLDBAC® cultures applied (Dalgaard et al. 2013b).

4. Discussion

4.1. Difference between attributional and consequential modeling

In LCAs of specialty ingredients, the difference between attributional and consequential modeling is often modest. But there are situations where the differences actually matter. These can be situations where a part of the product system that is constrained, is left out in the consequential model. An illustrative example is SFIs that enable a partial replacement of milk fat with palm oil in cream cheese. This results in a significant improvement when attributional modeling is applied, but has no effect when consequential modeling is used. The latter presumes, according to the consequential model we have used, that milk fat is a dependent co-product constrained by the main product milk. Hence, instead of influencing the production of milk, a change in demand for milk fat would affect the marginal oil on the world market (namely palm oil). In terms of modeling, this implies that we replace palm oil with palm oil – hence no effect.

4.2. Consequential modeling (pros and cons)

The advantage of consequential modeling is arguably that it better reflects the functioning of the market and attempts to model consequences of concrete decisions or choices in a concrete market context. From a consumer perspective, it could be the environmental impacts of buying product A versus not buying it – or buying product A instead of product B. Both situations imply changes – which is exactly what consequential LCA attempts to model. A hot-spot assessment is not necessarily about changes – but as soon as it is used to guide decisions or to make strategic choices it does concern changes. Consequential modeling is tailor-made to decision support and is better able to address the bigger picture including the direct and indirect market effects.

Consequential modeling avoids co-product allocation – often by system expansion. This leaves less room for arbitrary choices of allocation methods, but on the other hand consequential modeling opens up for different choices and assumption related to identification of marginal products and other market aspects. It could be argued that consequential modeling, by including markets aspects, requires too much from the LCA practitioner and makes LCA a too broad research discipline. Some would probably argue that ‘the best is the enemy of the good’ in this context. Ultimately one of the fundamental challenges of consequential LCA is that it can be hard to comprehend, not least for non-experts. As an example it can be difficult to communicate that an LCA of soy oil based emulsifier does not include soy oil in the inventory, because it is constrained by the demand for soy meal. The consumer, customer or other stakeholder might get confused – and skeptical when they are told that palm oil is included instead because this is the marginal oil on the world market. But this is rather a communication issue than it is a real disadvantage of consequential LCA. Another challenge is that it, by putting attention to the marginal, could lead to a missed opportunity to collaborate with the suppliers of soy oil to reduce upstream impacts. This does not have to be the case, but could be the outcome when results are interpreted by non experts.

4.3. Attributional modeling (pros and cons)

Attributional modeling is often described as a more normative approach with fewer considerations about the functioning of the market. Instead it follows traditional supply chain logic and it better reflects how current supply chains are physical connected. The fewer number of variables related to the market, also reduce to option for modeling related differences when comparing results from LCA studies. Attributional modeling is arguably easier understand, and from a supply chain logic, it makes intuitively more sense that soy beans are included in an LCA of soy oil, instead of the marginal vegetable oil (palm oil) as suggested in consequential modeling. It can be argued that, attributional modeling better reflects the way many consumers and NGOs attribute a responsibility on the suppliers in the actual supply chain. An example is when stakeholders avoid agricultural products than are produced in regions where deforestation is taking place – by sourcing from other regions may just reflect a shift of burdens via indirect land use changes, and eventually the effect on deforestation may be the same.

The main disadvantage of the attributional approach is that it includes constrained suppliers, i.e. the results of an LCA include emissions from suppliers that will not change their production as a consequence of a change in the demand for the studied product. Further, the attributional approach often involves different allocation methods for multiple product output activities. Apart from making studies more difficult to compare, this approach ignores the real effects of by-products and the allocated processes fail in some cases to maintain basic balances such as mass balance, carbon balance, energy balance etc.

It is out of scope in the present article to make a more elaborate description of pros and cons, but it should be clear that depending on the purpose of the LCA, both approaches have advantages and disadvantages and that it can be fruitful to provide both perspectives in LCA studies.

4.4. Other aspects of Sustainability

The present article has focused on environmental impacts, mainly the carbon footprint. But there are obviously many other elements of sustainability such as food security, health and nutrition aspects.

Several initiatives are taking place to broaden aspects of sustainability within our own supply chain. Emulsifiers and hydrocolloids can enable reductions of salt and fat (including trans-fat) in food products. Reduced calorie sweeteners can obviously reduce sugar levels, and products such as Litesse® can increase fiber contents of foods. Other examples are probiotic cultures, which can promote a healthy bacteria flora in the digestive system. Social sustainability aspects related to food are however complex to describe and include in a short article about SFIs, and it is highly context and culture dependent. In Mexico for example, milk competes on price with soda. This means that solutions to make milk more affordable is likely to have a positive effect on both food security and health – despite reducing the amount of milk solids on a per liter basis. The theme about what defines a sustainable and healthy diet has recently been described in a discussion paper by Garnett (2014).

5. Conclusion

DuPont N&H continuously seeks to reduce the environmental impacts from all life cycle stages including raw materials, transport, processing and use. It should be recognized, however, that environmental burdens associated with SFIs typically are small or insignificant in the final consumer products. This is due to the small amounts used (typically < 1% w/w) and the relative modest carbon footprint of 3-4 kg CO_2e per kg average SFI. The latter is close to the footprint of the products in which they are typically used.

More importantly, SFIs can make a significant difference in terms of reducing the footprint of the food products in which they are used. SFIs have the largest potential to leverage sustainability in the use stage where they can affect all stages of the customer’s value chain. Three groups of SFIs can be distinguished depending on how they influence the customers’ value chains:

- SFIs that influence the first stage of the customers value chain (raw materials) – e.g. by allowing replacement of animal based with vegetable based protein or fat.
- SFIs that contributes to improve customers manufacturing efficiency.
- SFIs that mainly address the last stages of the customer’s value chain, by enabling food products to stay fresh longer and thereby reducing food waste.

Food enzymes are an example of specialty ingredients that can address raw material substitution, processing efficiency and downstream food waste. Other groups of SFIs are typically less comprehensive in their functionality, but they can have the same potential to reduce impacts such as isolated soy protein replacing animal protein.

DuPont N&H has identified 70 solutions that allow for carbon footprint reductions in the customers value chain. LCA screenings of 40 solutions reveal that these solutions allow for reductions between 10 and 100 kg CO_2e per kg SFI applied. But there are also examples where 1 kg specialty ingredient can lead to reductions of more than 1000 kg CO_2e by reducing food waste.

Detailed third party reviewed LCAs have been applied for cradle-to-gate LCAs, while the use stage has been modeled through screening level LCA. Both attributional and consequential modeling is applied in all studies and generally results vary little. There are cases, however, where significant differences emerge, especially for dairy products. While these differences may appear inconvenient they show that methodological uncertainty can be significant in certain cases.

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Boosting grain yield by including leguminous bioenergy crops in the rotation – a life cycle approach

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ABSTRACT

Introduction of rotational grass/clover for anaerobic digestion in a grain production system was evaluated using life cycle assessment. Energy requirement, global warming potential and land use were compared in two scenarios; one cropping system including only cereals and another also including two-year grass/clover. Use of fossil fuels and greenhouse gas emissions were reduced substantially, mainly due to replacement of diesel with biogas, but also when digested grass/clover biomass replaced N mineral fertilizer. High N use efficiency throughout the production chain was required for a high replacement rate of mineral fertilizer. The residual effect from rotational grass/clover increases the yield potential and provides N for subsequent crops. Both these aspects should be included in LCA of break crops through evaluation of crop sequences rather than specific crops. We recommend that introduction of different rotational crops relying on N-fixation be evaluated as a future strategy for improving productivity and environmental performance.

Keywords: Crop rotation, multifunctional bioenergy crops, N-fixation, land competition

1. Introduction

Bioenergy production in the EU is currently based largely on annual oilseed, starch, and sugar crops (Pedroli et al. 2013). Using common crops for bioenergy production may perpetuate monoculture cropping and there is thus a need for more innovative production systems with multifunctional benefits. At present, only a small proportion of bioenergy feedstock comes from perennial ligno-cellulosic species such as miscanthus, willow, reed canary grass, and switchgrass (AEBIOM 2013). Introducing perennial crops which stay in place for a longer period of time decreases the flexibility and increases the investment costs, and might therefore be less attractive from the farmer's perspective (Gabrielle et al. 2014).

There has been a substantial increase in global crop production in recent decades, but there are now indications that the yield increase has started to slow down (Foley et al. 2011). Sustainable intensification of existing farmland is required to meet the predicted future increase in demand for food. Sustainable intensification is often discussed in terms of higher efficiency, as more product delivered per unit input of different resources (Smith 2013). Soil fertility is critically important for obtaining high yields, and strategies for maintaining and increasing soil fertility are thus essential components of sustainable intensification. Replacing monoculture with more varied cropping systems can improve soil structure, decrease the vulnerability to pests and diseases, and improve long-term productivity (Zegada-Lizarazu and Monti 2011). Introducing bioenergy crops with benefits for subsequent food crops in a crop sequence can thus improve current cropping systems, thereby mitigating negative land use impacts to a varying extent. Perennial or annual crops relying on symbiotic fixation of nitrogen (N) are of special interest as feedstock for bioenergy production from an environmental point of view. One reason is their potential to reduce the N mineral fertilizer requirement of the actual crop and of subsequent crops in the rotation. Another reason is their potential to increase yield in subsequent crops in both the short and long term (*e.g.*, Gan et al. 2003). According to a literature review examining cropping systems in different countries, mean yield benefits of up to 20% or more can be obtained when break crops are included in crop sequences (Kirkegaard et al. 2008). These multifunctional aspects are important to consider in assessments of crops relying on symbiotic N-fixation and in future policy development.

In this paper, we briefly describe how the introduction of perennial grass/clover in rotation with cereal crops affects subsequent food crops, using results from an LCA study conducted with Swedish data. The aims were to evaluate energy use, greenhouse gas emissions, and land use for grain production when rotational grass/clover is included for anaerobic digestion in a cereal cropping system, and to discuss how rotational aspects affecting yield potential can be included in life cycle assessments.

2. Methods

In the scenario study, life cycle assessment was used to evaluate two crop rotations: (i) a reference scenario including winter wheat-winter wheat-spring barley in a three-year crop rotation and (ii) a grass/clover scenario including two-year grass/clover followed by winter wheat-winter wheat-spring barley. The functional unit was 1 tonne of harvested grain, since the main focus was on how grain production was affected when introducing grass/clover for biogas production into the rotation.

The scenario study included only the operating phase. The assumed location was the county of Uppland in eastern Sweden. It was assumed that the grass/clover was digested and further upgraded to vehicle fuel quality, replacing diesel. The generation of biogas in the grass/clover scenario was accounted for through subtraction of the avoided burdens from the other activities in this scenario. The digested mixture of grass and clover replaced mineral N fertilizer otherwise used in cereal production. Band-spreading with trailing hoses was assumed for the digested grass/clover. In total, 73% of the N requirement of the cereals in the grass/clover scenario was met by digestate, while 27% was met by N mineral fertilizer. More details and references are provided in Tidåker et al. (2014), while yields and fertilizer rates are summarized in Table 1.

Special consideration was given to the effects of the grass/clover on subsequent cereal crops in the rotation. The cereal yield assumed in the grass/clover scenario was modified based on a compilation of numerous Swedish field experiments examining the residual effects of two-year grass/clover (Lindén 2008). Accounting for higher expected yield after grass/clover ley is also recommended in practical farming by the Swedish Board of Agriculture. Therefore, additional yield in the first and second year of winter wheat following grass/clover of 1000 and 400 kg ha^{-1}, respectively, was assumed. According to recommendations by the Swedish Board of Agriculture, the N delivered from the N-fixing clover to the subsequent crop could be expected to be 40 kg ha^{-1}. Changes in soil organic carbon in the two different crop rotations were modelled using ICBM (Andrén and Kätterer 1997).

Table 1. Assumptions in the two scenarios based on regional standard yield data, long-term field experiments, and guidelines for fertilization from the Swedish Board of Agriculture

	Yield (kg ha^{-1})	Fertilizer (kg N ha^{-1})[b]	Type of fertilizer
REFERENCE			
Spring barley	4300	80	Mineral fertilizer
Winter wheat	5600	135	Mineral fertilizer
GRASS/CLOVER SCENARIO			
Spring barley	4300	80	Digestate
Grass/clover[a]	7000		
Winter wheat I	6600	110	Mainly mineral fertilizer
Winter wheat II	6000	140	Digestate

[a] Grass/clover yield expressed in kg DM.
[b] Only N considered to be available for the crop was accounted for.

3. Results

The primary energy use was 1480 MJ in the reference scenario and -2900 MJ in the grass/clover scenario, *i.e.*, more energy was produced in the latter scenario than was required for grain production (Table 2). Replacing diesel with biogas was the most important factor explaining the difference, followed by the avoided use of mineral fertilizer in the grass/clover scenario.

The contribution to global warming potential (GWP) was 351 kg CO_2-equivalents for the reference scenario and -102 kg CO_2-equivalents for the grass/clover scenario. These figures included carbon sequestration in the cropping systems under study, but not indirect land use change. The replacement of mineral fertilizer in the grass/clover scenario decreased the emissions of greenhouse gases, but this gain was offset by the increased emissions due to more field operations being required and higher N_2O emissions in the grass/clover scenario.

The production of 1 tonne of grain occupied 0.20 ha in the reference scenario and 0.31 ha in the grass/clover scenario. The higher land requirement in the grass/clover scenario was due to the fact that the cereals occupied only 60% of the total area. However, due to the yield-increasing effect of the grass/clover, grain production in this scenario was 66% of that in the reference scenario, *i.e.*, higher than the proportion of occupied land.

Table 2. Primary energy use and global warming potential (GWP) for the two scenarios

	Reference scenario		**Grass/clover scenario**	
	Energy use	**GWP**	**Energy use**	**GWP**
Farm operation + transport	327	24	781	66
Mineral fertilizer production	1156	156	425	46
Direct field emissions		177		273
Indirect emissions		12		24
Gas production			1972	156
Carbon sequestration		-18		-216
Substitution of diesel			-6080	-451
TOTAL	1482	351	-2902	-102

4. Discussion

Bioenergy production based on dedicated crops grown on arable land has been increasingly questioned due to its negative land use impact. Competition between food, feed, and fuel crops is an important aspect to consider, especially since the demand for food is projected to increase substantially. This competition can be reduced or even eliminated for bioenergy crops by using residues or perennials from marginal land, or by growing additional crops between the main seasons or between rows (Tilman et al. 2009). However, bioenergy substrates are unlikely to be produced commercially on marginal land due to low economic incentives (Bryngelsson and Lindgren 2013) and residues for bioenergy production are limited. Rotational grass/clover could be another alternative, because it provides several benefits in a cropping system perspective, *e.g.*, N-fixation, carbon sequestration, *etc*. In this study, the improved short-term soil productivity due to crop sequencing effects was addressed by assuming higher yield potential for winter wheat in the first and second year after the grass/clover crop. It is likely that a long-term increase in productivity will also occur if the break crop recurs in the crop rotation, so assuming only a two-year effect might be somewhat conservative.

Different indicators for land use impact have been suggested, with soil organic matter proposed as a general indicator of soil fertility (Milà i Canals et al. 2007). Changes in soil productivity through increased yield potential can not only be measured as a separate indicator for land use impact, but can also be directly reflected in the area required for producing a certain amount of crops.

Higher yield potential for subsequent cereal crops following the grass/clover could not offset more than a small proportion of the displaced grain yield. However, the increasing yield potential together with carbon sequestration and N delivered from crop residues can counterbalance some of the negative impact from *e.g.*, indirect land use change, and is thus important to consider as a future strategy for mitigating climate change and decreasing the competition between food, feed, and fuel crops. It is therefore a need for environmental assessments of rotational crops relying on N-fixation in different cropping systems, taking into account their multiple benefits. One such crop besides clover is alfalfa (Gabrielle et al. 2014).

The N delivered to the wheat, as a residual effect from the grass/clover, replaces N mineral fertilizer in the subsequent crops. Applying a cropping system perspective and including the crop sequence rather than one specific crop is thus crucial to capture the multiple benefits. Digested mixtures of grass and legumes have the potential to replace a considerable proportion of the N fertilizer otherwise used in cereal production. Careful handling of the N-rich residue is crucial for many environmental aspects, and a sound management strategy is thus important. Experimental field data indicate large variations in ammonia volatilization (Webb et al. 2013). Anaerobic digestion increases the pH in the residue, which increases the risk of N losses. More experimental studies evaluating different biogas management systems are required in order to maximize the environmental benefits and minimize the emissions.

5. Conclusions

Introducing rotational grass/clover for anaerobic digestion into crop sequences can sequester carbon and replace a considerable proportion of the N mineral fertilizer otherwise required in cereal production. Furthermore, subsequent crops benefit from the residual effect of the grass/clover, thus increasing the grain yield potential. All these multiple benefits can help counteract some of the negative effects of indirect land use change and need to be included in assessment of leguminous crops for bioenergy production.

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Moving toward scientific LCA for farmers

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ABSTRACT

Certain complexities in the agricultural production differentiate this sector from the conventional industrial processes. The main feature to take into account is that the resources consumption and production is subjected to high variability in soil, rainfall and latitude. We show here an environmental tool developed in close cooperation with farmers in order to achieve applicability and comprehensibility. The procedure relies on the data retrieved from parcel management monitoring of different crops, mostly allocated in Catalonia and Ebro river watershed region (NE Spain). A set of comprehensive but also simple reports are provided, including material and energy balances, agronomic efficiencies and water and carbon footprints. The calculations also cover impacts due to infrastructure, including the estimation of materials in the use of greenhouses. Besides, a simple algorithm for reporting uncertainty using an approximation method of error propagation was added using the input uncertainties as defined by their data pedigree.

Keywords: decision making, calculation tool, sustainability indicators, growers, uncertainty analysis

1. Introduction

Today's society is more aware than ever of the state of the planet and the necessity of enhance the environment protection to deal with new issues such as global warming, overpopulation, water scarcity, etc. To face these challenges environmental regulations become more restrictive, and the productive sectors are forced to adopt measures, providing reliable information and applying tools for understanding and mitigating environmental damage. Particularly, the global food system, from land use change, through fertilizer manufacture, to food storage and packaging, is responsible for about one-third of all anthropogenic greenhouse gases (GHG) emissions (Vermeulen et al 2012). In this sense, the Global Warming Potential (GWP) is the most widely studied impact category as a result of the evidences of the rising global temperatures that are accompanied by changes in weather and climate thus affecting ecosystem and society. The GWP measures the environmental impact from a life-cycle perspective as the sum of GHG emissions caused by an organization, event or product, expressed in CO_2 equivalents. International standards such as the ISO 14067:2013 (ISO, 2013) provide guidelines for calculating carbon footprints. Furthermore, specific standards such PAS 2050.1 (BSI, 2011) provide specific guides for measuring the Carbon Footprint (CF) of horticultural and agricultural products.

Together with climate change, water scarcity has become one of the biggest environmental problems worldwide. The lack of access to fresh drinking water and sanitation has major impacts on people's well-being, causing massive health impacts. Moreover, the lack of water for productive purpose results in malnutrition, poverty and illnesses for a large portion of the world population. Agriculture represents a serious burden as the largest consumer of water and the main source of nitrate, ammonia and phosphate pollution in ground and surface water. It is of capital importance the implementation of good water management practices in irrigated agriculture, in an attempt to fix this increasing problem while providing food for the fast growing population. The concept of Water Footprint (WF) introduced by Hoekstra et al (2011) provides a framework to analyze the link between human consumption and the appropriation of the freshwater resources that incorporates both direct and indirect water use of a consumer or producer.

Nevertheless, the application of standardized LCA methodology to the evaluation of the agriculture sector has more complexities when compared to conventional industrial processes. The main issue to take into account is that the resources consumption and production is subjected to high variability in soil and weather conditions. Besides, the lower energy consumption of agriculture activities compared to industrial processes increases the relative significance of other parameters like transport or infrastructure. Therefore, in addition to the complexity of LCA tools we need to account for the extreme variability of agriculture. All these complexities and variability in the LCA indicators associated to agriculture make unpopular their application for farmers.

With the aim of involving the farmers in the tasks of accounting sustainability in their decisions and to retrieve from them reliable information, the principle of parsimony "as simple as possible and as complex as necessary" (Pidd 1996) is applied to develop the structure of a combined tool for the sustainable performance evaluation of agriculture processes.

Within the calculation and software tools available in the agricultural field, two calculators stand out: the Fieldprint calculator (www. fieldtomarket.org/fieldprint-calculator/) that assesses the CF and WF associated to crop production, and the Cool Farm Tool calculator (Hillier et al 2011) that is an open source software that considers crop management, livestock and manure management, field energy use and primary processing energy use. Both the mentioned software tools as well as the tool presented hereby are designed for farm-scale usage and for a certain growing season because of the variability above mentioned. Moreover, they are intended to be a decision-support tool for the farmer based on inputs that are well-known by the user.

Within the context of LCA, uncertainty analysis is used to better explain and support LCA conclusions based on the cumulative effects of uncertainty and variability. Reporting the outcome of the model with a quantitative measurement of the data quality means an added value compared with building deterministic models that assign single values to model parameters to obtain results as point estimates. In this sense, ISO 14044:2006 lists under "data quality" several aspects such as reliability, uncertainty/precision, methodological consistency, data sources used and reproducibility. The procedure implemented in the tool for the uncertainty quantification provides additional information about the confidence of the LCA results.

2. Methods

The calculations of the model are based on the primary data provided by the farmer at Farm Management Unit (FMU) level, which is the portion of land for which the data are representative. Two types of indicators are computed for FMU and growing season, carbon footprint (CF) and water footprint (WF), besides other agronomic parameters such as crop yield, gross irrigation requirements, pumping energy use efficiency and water used efficiency that are directly connected with the production costs.

The applicability of the tool is shown with four case studies: corn, nectarine, grape crops and tomato production in low-tunnel greenhouse. The obtained indicators are referred to mass of product (kg or t), although the reference flow for the compilation of the input data in the inventory questionnaire is referred to hectare and campaign. The use of this unit simplifies the work of the farmer when providing the information, because the amount of materials and energy corresponds to the spent for a hectare of land for the whole campaign assessed (a year in annual crops).

To carry out the CF and WF calculation a cradle to gate system boundary was considered. The frame includes the pre-farm processes such as the extraction of raw materials, production and transport of inputs. Within the production system (gate to gate) activities, the tool assesses the transport of the product to the closest cooperative, tillage tasks, energy consumption, irrigation, heating (protected crop), waste management, packaging and auxiliary equipment. Results of CF and WF are presented distributed in the different stages so the user can extract some valuable conclusions about the relative GHG emissions, material, energy and water consumption in the different process steps, for future environmental improvements and savings.

The software is developed in a spreadsheet format for its adaptation as a web tool with the following structure:

- Inventory questionnaire: list of necessary activity data for the compilation of all model inputs that have to be provided by the user.
- Database and default data: secondary data are provided from datasets of Ecoinvent Database (http://www.ecoinvent.org/), likewise default data for intermediate calculations are included (such as life span or material composition).
- Greenhouse module: equations for the accounting of materials involved in the greenhouse infrastructure.
- Calculation module: translates the activity data that characterize every sub-process in the system boundaries to environmental impact indicators considered.
- Uncertainty analysis module: applies an approximation method of error propagation from the activity data and emission factors uncertainties.
- Results: numerical and graphical report of the resulting values.

2.1. Inventory and scope

The system boundary was defined from raw material extraction to farm gate. Pre-farm processes (often referred to as 'cradle') such as the extraction of raw materials, production, and transport of inputs used on the farm are also included in the assessment. The production system (farm-gate) is structured in different stages: fuel, transport, energy, fertilizer, treatments, packaging, infrastructure, etc. Fuel stage includes the fuel consumption from farm tasks, and energy stage involves the pumping consumption for irrigation. The transport stage includes the production transport from farm to wholesale, the transport of materials within the farm and the transport of waste materials to a treatment plant. The fertilizer stage implies both manufacturing and consumption. In addition, four types of greenhouse structures are available: glass, multi-tunnel, "parral" or "Almeria-type" and low-tunnel. The assumptions applied for the average dimensions and the materials involved in each type can be found in Antón et al (2013), as well as the equations that relates the size and other parameters (depending of the type and subtype of greenhouse) with the amount of materials used.

2.2. Carbon footprint

The carbon footprint is a measure of the impact that human activities have on the environment in terms of the amount of GHG emitted over the full life cycle of a process or product measured in units of carbon dioxide equivalents ($CO_{2\text{-eq}}$). The CF is calculated based on the IPCC guidelines (IPCC 2006) that suggest a time horizon of 100 years for the decay rates and can be expressed as follows (Eq.1):

$$CF = EV * AD \qquad \text{Eq. 1}$$

where, *EV* is the emission vector per unit of reference for the specific activity (*e.g.* kg $CO_{2\text{-eq}}$ per kg of pesticide for the activity of pesticide manufacturing); *AD* is the activity data that expresses the intensity of the activity in the units of reference per functional unit (*e.g.* kg of pesticide$\cdot$(ha$\cdot$y)$^{-1}$).

The calculations include the N_2O induced emissions from managed soils, both direct and indirect releases. It comprises the emissions due to nitrification and denitrification processes from soils to which the nitrogen is added, volatilization of NH_3 and NO_x and subsequent redeposition, and leaching and runoff of nitrogen. Default emission factors are used for the evaluation of these three pathways based on the equations proposed in the chapter 11, volume 4 of the IPCC guidelines (IPCC 2006).

2.3. Water footprint

The water footprint indicator distinguishes the quantity of water consumed using a color code (green, blue and grey) depending on the type of water sourced and polluted. WF in a crop can be calculated using equation 2 (Hoekstra et al 2011):

$$WF = WF_{green} + WF_{blue} + WF_{grey} \qquad \text{Eq. 2}$$

where, WF_{green}, is the volume of rainwater consumed (the total rainwater evapotranspiration plus the water incorporated into the harvested crop); WF_{blue}, is the volume consumed of fresh surface or groundwater; WF_{grey}, is the volume of freshwater required to assimilate the load of pollutants based on natural background concentrations and existing ambient water quality, although this concept was left out of the scope in this study.

The mentioned water volumes can be used as translated to midpoint and endpoint impacts in assessment methods when applied to freshwater consumptive use by using the Water Stress Index (WSI) (Pfister et al 2009). The WSI can range from 0.01 to 1 and indicates the portion of consumptive water use that deprives other users of freshwater for a certain watershed, where a value of 1 means a serious water stress in a basin (0.259 for the Ebro basin). Therefore, the water footprint impact assessment for WF_{blue} and WF_{green} can be calculated as follows (Eq. 3 and 4):

$$WFIA_{blue} = WF_{blue} * WSI \qquad \text{Eq. 3}$$

$$WFIA_{green} = dGW * WSI \qquad \text{Eq. 4}$$

where $WFIA_{blue}$ and $WFIA_{green}$, are the water footprint impact assessment for the WF_{blue} and WF_{green}, respectively; dGW is the delta green water consumption (Nuñez et al 2012) that can be calculated as the difference between the WF_{green} and the consumed by the reference system (crop evapotranspiration) represented as ET_c.

2.4. Uncertainty analysis module

The procedure included in this module assesses the uncertainty that arise from two types of parameters: activity data and characterization factors. The approach follows the detailed guidance of the ILCD Handbook about data quality concept and approach and the data quality guidelines for Ecoinvent database v.3 (Weidema et al 2012).

Parameter uncertainty is represented by a lognormal probability distribution using the Pedigree Matrix approach that relates quality indicators to uncertainty ranges. Both basic and additional uncertainty, through variances of the underlying normal distribution, can be assigned. On the one hand, the activity data uncertainty is defined by the user/farmer following quality rules depending on the type of data estimation. However, there are available default uncertainty values according to different types of exchanges. On the other hand, basic uncertainty of the characterization factors is quantified for each LCA vector included in the inventory, as well as additional uncertainty according to five independent characteristics: reliability, completeness, temporal correlation, geographical correlation and further technological correlation.

As long as the tool needs a prompt response about the environmental performance of the scenario (i.e., for a given growing season and facility), an analytical propagation method is proposed. Particularly, Taylor series expansion method (Ciroth et al 2004) was formulated in the model to retrieve the uncertainty of the studied scenario computing the geometric standard deviation (GSD) as a function of each parameter GSD and its sensitivity associated (i.e., the influence or contribution of the impact due to a certain parameter over the total inventory impact). Therefore, the impact indicators of the crop (e.g., kg $CO_{2\text{-eq}}$ per kg of product) for a given season and field are depicted with an error bar.

3. Results & discussion

Figure 1 shows the results that the tool provides in terms of the carbon footprint impact for the production during a campaign of corn, grape and nectarine, with yields of 50800, 15300 and 47100 kg·(ha·y)$^{-1}$, respectively. The outputs are detailed for different productive processes considered in the system. Besides, error bars are included to represent the uncertainty associated with each value. From the results of the carbon footprint is possible to conclude that the contribution of fertilizer production and application to the global warming potential is the highest in grape and corn crops, while for nectarine the fuel consumption has the highest impact.

In addition, the uncertainty analysis yields the worst value for the corn production while the lowest uncertainty is obtained for the vineyard farm. These outputs are explained by the influence that the most uncertain activities have in the overall result. It is the case of the impact for transport and the use of fertilizers. In the case of transport, the uncertainty is due to the default uncertainty associated to the activity data. In the case of fertilizers the emissions factors have high uncertainty according to their data pedigree, as well as the high default uncertainty associated with the emissions of N_2O for the nitrogen application. Both activities are determinant in the corn faming because of their high contribution in the total impact.

In Figure 2 the water footprint values obtained for the three mentioned crops are displayed. The results raise higher water footprint impact assessment for corn production due to a higher value of blue water footprint considering the Ebro basin, followed by the grape and finally the nectarine production as the less water consumptive. The same trend for the green water footprint is shown, that is also higher in the case of corn farm.

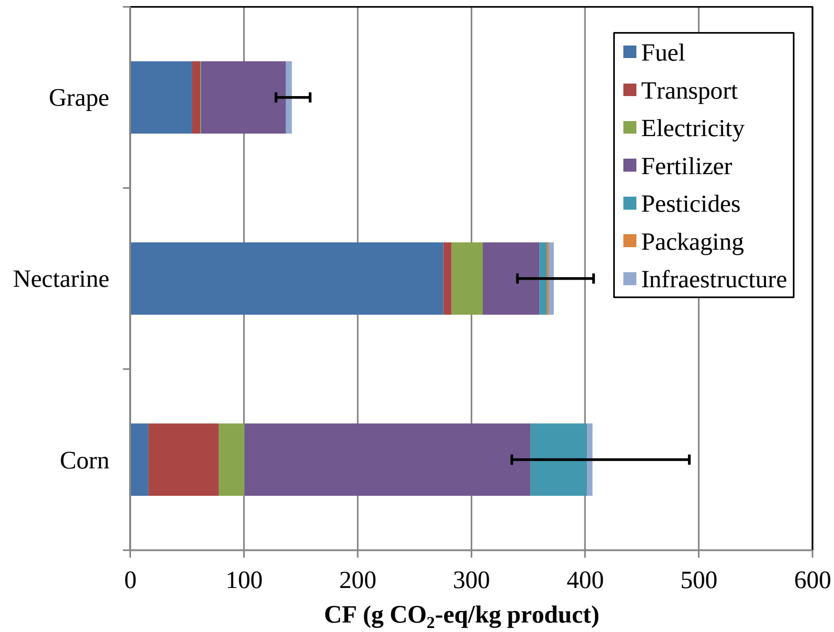

Figure 1. Carbon footprint results for each type of crop and distributed in different activities of productive process.

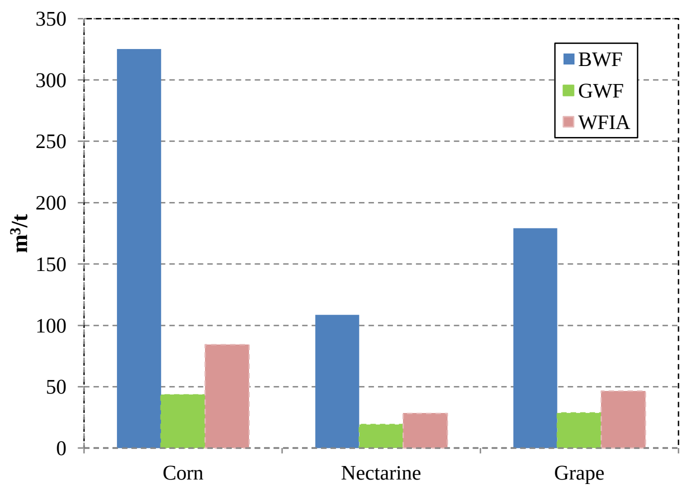

Figure 2. Blue and green water footprint (BWF and GWF) results and the total water footprint impact assessment (WFIA) for each type crop.

The production of 150000 kg·(ha·y)$^{-1}$ of tomatoes in a low-tunnel greenhouse without heating was evaluated obtaining a total carbon footprint of 293 g $CO_{2\text{-eq}}$/kg of tomatoes. The water footprint was not analysed for this time. Figure 3 shows the share of the different activities included in the inventory. The carbon footprint calculation for the tomato production in low tunnel greenhouse highlights the importance of including the infrastructure impact in the protected crops evaluation. Particularly, near a 30% of the global warming potential impact comes from the greenhouse structure followed by the fertilizers contribution. Nevertheless, the contribution of the structure would to the total impact would be reduced the 10% if the crop requires climate control systems since their impact usually become the most important.

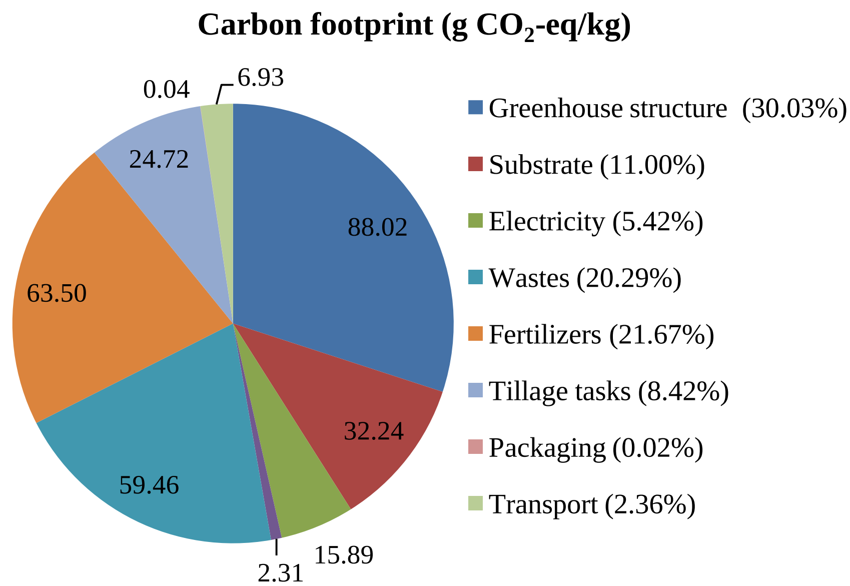

Figure 3. Carbon footprint impact for the tomato production in low tunnel.

Table 1 shows the values reported for the uncertainty analysis in the case of the corn production. The GSD of the whole inventory allows us defined 95% confidence interval for the calculated environmental indicator; particularly, the values correspond to the carbon footprint calculation. The Monte Carlo simulation was applied for the internal validation of the analytical propagation method proposed. More precisely, 5000 iterations were simulated using this method obtaining the corresponding confidence interval and the relative error indicated in the fourth column. A relative standard deviation of 24% was obtained from the uncertainty propagation procedure over the CF value obtained for corn production. The selected approximation method yields similar results to the Monte Carlo simulation with a difference lower than 5%, although the stochastic procedure tends to overestimate the impact. Nevertheless, the uncertainty outputs can be qualitatively characterized by comparing the resulting uncertainty with the intrinsic variability of the process, which is due to several factors varying with growing season and farm characteristics (e.g., crop production, weather conditions, soil conditions, etc.). Therefore, a quality rating can be derived depending on where the scenario's uncertainty is located with respect to the intrinsic variability of the system. This can be accomplished taking advantage of the potential spreading of the use of this tool, allowing us the access to primary data from different estates, crops and weather conditions.

Table 1. Results reported by the uncertainty analysis module for the carbon footprint of corn production.

	Taylor series expansion approximation	**Monte Carlo simulation**	**Error %**
Geometric mean	0.407	0.419	3.568
Geometric standard deviation	1.100	1.108	0.708
95% lower bound	0.334	0.342	2.187
95% upper bound	0.490	0.516	4.930

Some innovative aspects of the presented tool can be highlighted: the adaptation of the required data for the LCA to the reality of farms after conducting a sensitivity analysis for the identification of key inputs; the linking between the tool and quality standards for processes that are implemented at farm level, such as Nature 's Choice, GLOBALGAP or Integrated Production; and the best available technology for the data acquisition in surveillance systems to measure and calculate the actual values of the parameters needed for the evaluation and that can be also used to compute the effects of mitigation measures.

4. Conclusions

The developed tool provides a thorough inventory to assess the carbon and water footprint at farm level that is at the same time accessible and handy for the farmers to make available the reliable primary data needed. The tool is designed to support decision making, adapting the requirements of data to the reality of the farms.

The application of life cycle assessment indicators (carbon and water indices) enabled to take advantage of their benefits for suggesting environmental improvement in crop production. Through the carbon and water indices is possible to identify hotspots within the studied life cycle stages such as tillage task, energy consumption, treatment, fertilization, transport, etc. Moreover, the feedback from the users and stakeholders helps to incorporate new features and to continuously improve the tool.

The findings of this work reveal that the use of fertilizers has an important contribution in the global warming potential and hence it can be a priority in the reduction of the greenhouse gas emissions. Besides, the use of fertilizers is a source of uncertainty; therefore it is vital to have an accurate estimation of the fertilizer's contribution by means of the collection of high quality primary data about the chemical amount and composition, and the use of recognized emissions factors for the fertilizer manufacturing. The module for the calculation of the materials of the greenhouse structure is a value add-in given the widespread use of protected crops and the high impact in terms of carbon footprint of which greenhouses structures are responsible.

Additionally, the procedure for the uncertainty quantification module provides additional information about the confidence that the LCA inventory results can have. The method is intended to improve the agricultural activities management. The impact indicators accompanied by the confidence interval derived provide an important perspective when interpreting the results and when defining benchmarks for product comparison.

To achieve the indispensable participation of the farmers in the compilation of the inventory data, their cooperation is rewarded by raising awareness of the benefits that the results of the resulting reports can offer: adjusting the resources consumption (water, fertilizers, energy requirements, etc.); application of simple metrics for the diagnosis, comparison and impact evaluation of the management decisions; useful information in accordance with other quality requirements (e.g., Global GAP); and the availability of technical arguments for the product distinction in the market. Moreover, farmers use the tool based on data from their usual field notebook, in this way they can directly relate the resulting indicators, such as water or carbon footprint, to their common practice. Thus, a reduction in diesel consumption in irrigation and machinery can be evaluated simultaneously as economical and environmental savings.

Furthermore, the benefits are extended to the supply chain and stakeholders since the results allow them to evaluate different suppliers based on the mentioned metrics by product origin, thus helping in a better communication with the final consumer.

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Climate friendly dietary guidelines

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ABSTRACT

The aim of this study was to investigate how the present Danish diet could be changed in a climate friendly direction that follows the recommendations of a healthy diet. The carbon footprint (CF) of an average Danish diet was calculated and compared to CF of a recommended healthy diet by 1) modifying the average diet according to the Danish food based dietary guidelines, 2) and adjusting to ensure an iso-energy content and a nutrient content according to the Nordic Nutrient Recommendations. Afterwards the healthy diet was changed further to reduce CF.CF from the diet was reduced by 4%, if the healthy diet was eaten instead of the average current diet. However, if the diet was climate optimized by choosing foods with a low CF within the food groups; meat, vegetables and fruit, CF of this diet may be reduced by 23 % compared to CF of the average diet.

Keyword: Danish average diet, carbon footprint, Danish food based dietary guidelines

1. Introduction

In recent years focus has been on emission of greenhouse gases (GHG) from the food production. It has been estimated that the food sector in the developed countries contributes with up to 30 % of total GHG emission (Heller et al 2013; Tukker et al. 2009). Several Climate Summits organized by United Nation (latest COP19 was held in Warsaw, Poland from 11 to 23 November 2013) have been held in order to address the climate change at an international level, which unfortunately didn't result in specific targets or actions (UN 2009; UN 2010).

The special thing about food production is that in addition to contributing to emission of CO_2 from fossil energy consumption it also contributes with the so-called non-energy-related GHG emission – in form of the nitrogen oxide (N_2O) emissions related to use of fertilizers, deforestation (CO_2), and methane (CH_4) from ruminant digestion.

The diet and consequently the related food production contribute to the GHG emission. The different stages of each food items life cycle contribute to the GHG emission: the primary agricultural food production, food processing, as well as during transport and storage of food (Carlsson-Kanyama et al. 2003; Carlsson-Kanyama and Gonzales 2009; Garnett 2008; Garnett 2011; Nielsen et al. 2003). Finally, also the cooking at home contributes to the GHG.

The climate impact can be assessed from different perspectives; a production perspective i.e. the burden of the total food produced in Denmark inclusive emissions from food that is exported to other countries, or alternative from a consumer perspective – i.e. the burden of the food consumed in Denmark exclusive emissions from food that is exported and inclusive emissions from food that is produced outside Denmark. Seen from the production perspective, Denmark's agricultural production was estimated to represent 16 % of the total Danish emission of GHG (Olesen 2008). The climate impact of the Danish food consumption from the consumer perspective was calculated to be around 2.8 tons CO_2-eqv / person/year – equivalent to approximant 15.4 mill. tons CO_2 for the total Danish population and thus approximate 25 % of all GHG from The Danes' total consumption according to IDA's Climate Plan 2050 (Anonymous 2009).

Different types of foods contribute to different degrees to the climate impact. There are big differences between the level of GHG emission from different food groups e.g. between animal products such as meat and cheese and vegetable products such as vegetables, flour and grain. Also within the various food groups there are differences, e.g. between different types of fish or meat depending on the way the products are produced (Carlsson-Kanyama and Gonzales 2009; Olesen 2009).

The amount of GHG emitted by a produced food product is called the food's carbon footprint (CF). CF or global warming is one among several impact categories; acidification, nutrient enrichment, photochemical smog and land use etc. The CF is calculated by a life-cycle assessment (LCA), which includes GHG emission from the foods whole life cycle: agriculture, horticulture or fishing, including emissions related to the production of inputs such as fertilizers, processing, transportation and storage of food products until the food products are placed on the shelf in the supermarket. The CF of the preparation at households is typical not included (Mogensen et al. 2009a; Garnett 2008) and not needed in some comparison studies (Tukker et al. 2011). Some LCA calculations also include estimations of GHG contribution derived from food waste (Mogensen et al. 2011; Anonymous 2009). Differences in the used LCA calculation methods may complicate comparisons between CF from different sources. Furthermore, production methods and energy resources change over time, and therefore influence the results of calculations.

This study (Thorsen et al. 2012) was based on Danish food based dietary guidelines from 2005 (Astrup et al. 2005) including currently updates. They aim at increasing the intake of fruit and vegetables, bread and cereals (coarse or wholegrain) and fish, and a decrease in intake of fat from dairy and meat products and of sugar containing products. In 2013 the scientific evidence of the Danish food based dietary guidelines was updated. The main conclusions were that the increase of fruit and vegetables was maintained, fish intake was increased, an increased intake of wholegrain cereals was specified and the decrease in fat from dairy and meat products was emphasized. Furthermore the intake of red meat (cow, sheep, pig) and the sugar containing foods were restricted, especially sugar containing beverages.

1.1. Objectives

The aim of this study was to investigate how the present Danish diet could be changed in a climate friendly direction and at the same time following the recommendations of a healthy diet.

2. Methods

The carbon footprint (CF) of the average Danish diet estimated from the National Danish Survey on Diet and Physical Activity from 2003 to 2008, including 3354 adults, 18-75 years of age, hereof 47% men (Pedersen et al. 2010) was estimated. CF of an average diet was compared with CF of a diet that follows the Danish dietary recommendations. This diet was obtained by 1) modifying the average diet by scaling the food groups according to the Danish food based dietary guidelines (Astrup et al. 2005), and 2) adjusting to ensure an iso-energy content of the diets and a nutrient content that follows the Nordic Nutrient Recommendations (Nordic Council of Ministries 2004). The healthy diet was designed by using a modeling tool developed from the Danish nutrient calculating system GIES (Biltoft-Jensen et al 2008).

To assess the impact of food production on global warming the CF from each food item was calculated by use of life cycle assessment (LCA) (ISO 2006a; ISO 2006b). The LCA method implies that all emissions of GHG from cradle to grave are included. I.e. all GHG emissions from production and transportation at the farm, but also the GHG emissions related to processes after the food leave the farm are included. The CF from individual foods i.e. is expressed in CO_2 equivalent per unit of food produced, and the functional unit (FU) is one kg food. The CF values are from Mogensen et al. (2009 b) and are based on data from Nielsen et al. (2003). The average CF for the individual food group is weighted according to the distribution of different types of food items within the food group in both diets.

In the calculation of the CF of food intake the contribution from GHG emissions related to food waste was taken into account from the whole food chain, from production to retail and households. As there is no Danish estimations of food waste in households, waste was estimated from an English report about food waste on individual food groups; on average about 20% of purchased food ends up as edible waste, i.e. waste that could have been avoided (WRAP 2008).

The healthy diet was investigated further with the aim to reduce CF of the diet by using the “hot spot” approach. Hot spot food groups were in focus, i.e. food groups where it is easy and convenient to change between different food products in the group in order to minimize the CF. Hot spot analyzes are interesting because in that way you could help the consumer to choose more climate friendly food, while taking into account also the Nordic recommendations (Nordic Council of Ministries 2004) and the 8 Danish dietary guidelines (Astrup et al. 2005).

When the hot spot perspective is related to the dietary guidelines it is important to focus on food groups that have a high food consumption (large amount of food) and/or include foods having a large CF (high CO_2/kg food). The next step is to assess how foods of the hot spot food groups could be exchanged with similar foods within the group with lower CF.

Finally, the Danish food based dietary guidelines from 2013 (Danish dietary guidelines 2013; Tetens et al. 2013) were investigated and if relevant, complementary advises related to improving the climate impact were added.

3. Results

The estimated CF from the Danish' average diet (2003-2008) and a modeled recommended diet that meets the dietary guidelines and the nutritional recommendations are shown in Table 1.

Table 1 comprises 14 different food groups. Both diets are scaled to an energy intake of 10 MJ/day. These food intakes also give rise to food waste in retail, processing and household. Regarding fruit and vegetables also a 20 % peel waste (inevitable waste) is included. Table 1 shows an estimate of how much the changes from the average diet of the modeled recommended diet would benefit the climate. It is seen that the largest differences obtained by a reduction of the amount of meat with approximate 50 g. This is to some extent cancelled by the contribution from the foods replacing the meat such as the food groups of fruit, vegetables, milk and fish which according to the Danish food based dietary guidelines 2005.

The calculation shows that CF from the diet was reduced by 4%, if the healthy diet was eaten instead of the average current diet. The largest reduction is a result of reduced intake of meat and beverages (beer and wine). The GHG contribution of beverages (excluding milk and juice) has declined by almost 40 %. Overall the recommended diet has a CF that was approximate 4 % lower than the current average diet.

Table 1. The carbon footprint (CF) for an average Danish diet and for a modelled recommended diet (that fulfills both the dietary guidelines and the Nordic recommendations). Both diets are scaled to a daily energy intake of 10 MJ. The CF values are from Mogensen et al. (2009 b) and are based on data from Nielsen et al. 2003.

Food group	CF, kg CO_2/kg food	Intake g/person/day		Food waste, % [6)]	CF form the diet gram CO_2/person/day	
		Present average diet	"Modelled recommended diet"		Present average diet	"Modelled recommended diet"
Milk, dairy products	1.2	359	500	3;2;2	462	644
Cheese	11.3	38	25	3;2;2	455	303
Bread, rice, pasta	0.8-3.3	236	274	31;6;2	403	480
Vegetables	0.1 – 2.9	186	304	(19-45);6;2; (0-20)	381	567
Potatoes	0.2	113	192	19;2;2;20	31	52
Fruit (ex. Juice)	0.4	245	271	26;6;2;20	230	260
Juice	1.0	80	50	18;2;2;0	102	63
Meat	3.6 – 19.4 [2)]	121 [3)]	87 [4)]	13;2;2;0	1599 [7)]	1277 [7)]
Poultry	3.4	27	31	13;2;2;0	149	171
Egg	2,0	19	25	18;2;2;0	49	63
Fish	1.8 – 10,5	25	42	13;2;2;0[8)]	170	292
Fats	5.1	38	32	3;2;2;0	219	173
Sugar and candy	1.0	36	23	18;2;2;0	41	29
Beverages	0.02 – 2.1	2273	1955	3;2;2;0	698	417
Total diet [1)]					**4986 (100)**	**4790 (96)**
Recommended and climate friendly diet [1,5)]						**3864 (77)**

1) Beverages are included, figures in brackets: percentages related to average diet.
2) Climate footprint per kilo carcass – Amount of carcass behind an intake is calculated by using a factor 1,47 for beef, 1,33 for pork and 1,38 for poultry (chicken)
3) Type of meat in the present average diet: Men: 135 g meat/day: 25 % beef, 75 % pork. Women: 106 g meat/day: 28 % beef 72 % pork.
4) Meat in the recommended diet: Men: 92 g meat/day: 30 % beef, 70 % pork. Women: 81 g meat/day: 33 % beef, 67 % pork.
5) Climate friendly diet: The only difference from "recommended" diet is: within the food group: fruit, vegetables and meat a climate-friendly solution is chosen e.g. For the food group vegetables carrot is chosen, for the food group fruit apple is chosen and for the food group meat the reduction is done for beef and then for pork.
6) A larger food production is needed than the food intake figures show since food waste is found in all parts of the food chain. The figures cover edible food waste in household ; retail; processing; and peel/skin (not edible, inevitable waste for fruit, vegetables and potatoes).
7) Incl. calculation from carcass to meat.
8) All fish products are calculated without bone like fillet and peeled shrimp

In Table 2 we take a closer look at the CF for the hot spot food groups: meat, vegetables, fruit and fish. From Table 2 it can be seen that it is possible within the different food groups to reduce the contribution to CF by choosing foods with a lower CF. In order to choose climate friendly foods it is better to choose e.g. poultry or pork instead of beef, and fruit or vegetables in season instead of food produced in a greenhouse or imported. Regarding fish a climate friendly choice would be herring or mussels instead of flatfish or shrimps.

In addition the diet was optimized to reduce CF of the diet by choosing foods with a low CF within the food groups; meat, vegetables and fruit, CF of this diet may be reduced by 23 % compared to CF of the average diet.

Table 2: Effect on the contribution to carbon footprint (CF) from different food groups dependent on choice of food item within food group

Food group	Foods item	"Modeled" recommended daily intake in gram[2)]	Food waste % [5)]	CF from the food items kg CO_2/kg food	Contribution to CF from the production of the daily intake of food [1)], gram CO_2/day
Meat and meat products	Average diet – mix of food items within the food group	87	13;2;2;0	9.2 [4)]	1277[6)]
	Beef	87	13;2;2;0	19.4 [3)]	2966 [6)]
	Pork	87	13;2;2;0	3.6 [3)]	498 [6)]
	Chicken, fresh	87	13;2;2;0	3.1 [3)]	445 [6)]
	Chicken, frozen	87	13;2;2;0	3.7 [3)]	531 [6)]
Vegetables (ex. potatoes)	Average diet – mix of food items within the food group	304	(19-45);6;2;(0-20)	1.32 [4)]	567
	Carrot	304	19;6;2;20	0.122	62
	Onion	304	19;6;2;20	0.382	195
	Greenhouse vegetables	304	19;6;2;0	2.7	1099
Fruit	Average diet – mix of food items within the food group	271	26;6;2;(0-20)	0.52 [4)]	260
	Orange	271	26;6;2;20	0.7	347
	Banana	271	26;6;2;20	0.5	248
	Nuts, almonds	271	26;6;2;0	0.88	350
	Danish apple, pear	271	26;6;2;20	0.1	50
	Imported apple, pear	271	26;6;2;20	0.4	198
Fish and fish products [7)]	Average diet – mix of food items within the food group	42	13;2;2;0	5.7 [4)]	292
	Herring, fillet, peeled, frozen	42	13;2;2;0	1,8	90
	Shrimp, frozen, peeled	42	13;2;2;0	10,5	528
	Codfish, fillet, frozen	42	13;2;2;0	3,2	161
	Flatfish, fillet, frozen	42	13;2;2;0	7,8	392

[1)] Calculation of CF is based on produced amount of feed, taking into account food waste.
[2)] Food intake is an average for men and women.
[3)] CF of meat is given as CF per kg carcass, needed amount of carcass per kg meat intake: 1.47 for beef 1.33 for pork and 1.38 for poultry.
[4)] Average CF for all type of meat in the meet group, weighted according to distribution of different types of meat, accordingly average CF for the other food groups
[5)] Food waste in household; retail; processing; and peel/skin (not edible, inevitable waste for fruit, vegetables and potatoes)..
[6)] Incl. calculation from carcass to meat.
[7)] All fish products are calculated without bone like fillet and peeled shrimp

In the climate friendly diet, we have reduced the meat intake by 50 g/day, removed the beef and then reduced the pork. The intake of vegetables was increased with 300 g/day, and the vegetables with the lowest CF (such as carrots) was chosen, and regarding fruit the amount of fruit was increased by 50 g/day and fruit with the lowest CF (such as Danish apple) was chosen. The results are shown in Table 1 above and in footnote 5.

4. Discussion

The present study showed that the climate impacts from human food consumption can be reduced by conscious food choices. Beside that minimizing food waste and choosing a more sustainable food production in relation to agriculture could reduce CF from food production further.

In this study, the most recent dietary data were used for rough calculations of the climate contribution of a recommended diet compared to the current Danish diet. According to our calculation the climate contribution from the diet will be 4 % reduced, if the recommended diet is eaten instead of the average current diet. A saving in carbon footprint in the order of 4% CO_2-eq is so small compared to the uncertainty of the data included that is not necessarily a real saving.

In addition to eating a recommended diet people would optimize their diet in a more climate friendly way by choosing foods with a low carbon footprint, especially in the food groups; meat, vegetables and fruit. Our calculations show that the CF of such a climate friendly diet including beverages would be reduced by 23 % compared to CF contribution of the average diet. Thus, a climate optimized recommended diet would provide a significant reduction of the CF (23%) as compared to a recommended diet (4%). The number (23 %) is not essential, it can be less or more, but the savings are significant compared to the recommended diet. In real life the savings from the food groups meat, fruit and vegetables probably would be less than estimated here, but on the other hand optimized choices within the other food groups is expected to provide further savings. Thus there is a great potential for reducing CF of the diet by choosing climate friendly within a recommended diet. If households further reduce their food waste, it will have a major effect on climate impacts. The total food waste in households is estimated to be around 20%, accounting for 12.5% of the CF of food production. However, the calculations are based on English data from investigating food waste in households (WRAP 2008). More recent data from Danish households are needed to validate this part of the calculation.

Other studies (Mogensen et al. 2009b; Saxe et al. 2006; Tukker et al. 2011) find similar to the present study that a diet following the dietary guidelines will have a slightly lesser impact on the climate all other things being equal. However, it has also been found that high-nutritional diets had significantly higher GHG emissions than low nutritional quality diets (Vieux et al. 2013), Another study found that the GHG emission was 27 % lower when the diet (the New Nordic) was climate optimized by choosing either less beef or by substituting all meat with legumes, dairy products and eggs (Saxe et al. 2013).

In this study the energy and nutrient content are estimated for the average as well as the recommended diet. The diets are comparable since the energy content equals, and the recommended diet is also adjusted so that the nutrient contents are in accordance to the recommended levels. (Vitamin D is an exception, so other sources should be taken into account, such as supplementation and/or production within the body exposing the skin to sunlight). Therefore the functional unit kgCO_2/kg food of CF is used for the foods and g CO_2/day for the diets. However, the present study is an approximation with regard to the CF calculations and is conducted to get an overview of the CF from the overall diet. More accurate calculations of the climate-optimized diet require more in-depth investigations and further nutritional calculations that are beyond the aim of this study. For instance in the climate-friendly diet the milk intake is set to be 500 g/day, although this could be reduced to 250 ml/day in accordance with the Danish dietary guidelines from 2013 (Tetens et al. 2013), it might not be in accordance with removing all of the beef. Furthermore, a decrease in milk content of the diet needs to be accompanied by an increase in other food items that will provide energy and nutrients in approximately the same amount.

Fruits, vegetables, cereals and potatoes, which according to the dietary guidelines should be by far the largest part of the diet, are all low in climate impact, especially when choosing open field grown products and avoiding products transported by airplane. Meat and cheese are generally high in CF, but CF from meat as beef and lamb is higher in CF than pork and poultry. Also vegetable oil should replace butter and hard margarines, vegetable oils generally have a lower CF. Stimulants as sweet and alcoholic drinks, sweets and cakes, which should decrease in the Danish diet, probably have a rather high CF, but the data concerning CF of this food group is weak and should be improved considerably. An increased intake of fish that is advised by the dietary guidelines would by all means increase the climate impact and might be problematic if leading to overfishing (Clonan et al. 2013), but the negative impact may be limited by a conscious choice of fish products. Eating the recommended diet would change the diet in the direction of lower fat and higher fiber content, e.g. by reducing the intake of red meat and cheese and instead eat more coarse vegetables and fruit, bread and grains. By further choosing foods with a low carbon footprint whenever possible the climate impacts of the food consumption is reduced significantly. The climate friendly additions to the dietary guidelines should be short advises that are easy to understand and follow by the consumer. Therefore advises regarding the transportation of the foods are not included, since it is very complicated to explain the exceptions and since the consumer usually has no knowledge about how and how far the food items have be transported. These qualitative advices was in line with guideline development in Sweden and the Netherlands (Fogelberg 2008; Health Council of the Netherlands 2011).

5. Conclusion

It was concluded that there is potential synergy between a healthier diet and a more climate friendly diet. There seems evidence to complement the Danish dietary guidelines 2013 with the following advice *(in italic type)* to reduce the CF from the diet:

- Eat a varied diet, not too much, and be physical active
- Eat fruit and many vegetables – *preferably open field and in season*
- Eat more fish - *choose the climate friendly fish; herring and mussels*
- Choose whole grain
- Choose low-fat meat and meat products – *choose pork and poultry rather than beef and lamb*
- Choose low-fat milk and milk products - *restrict intake of cheese*
- Eat less saturated fat - *choose vegetable oils rather than animal fat*
- Eat foods with less salt
- Eat less sugar - from soft drinks, sweets and cakes
- Drink water - rather than sweet and alcoholic drinks
- Avoid overeating - *and waste - will also reduce CF from food production since less food need to be produced*

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Carbon footprint calculator for European farms: preliminary results of the testing phase

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ABSTRACT

To facilitate the adoption of low-carbon farming practices in Europe, the European Commission developed a carbon calculator that assesses greenhouse gas (GHG) emissions and recommends mitigation actions suitable for each farm. The Carbon Calculator, which quantifies GHG emissions based on international standards for life cycle assessment and carbon footprinting, delivers carbon footprint results both at the farm and product level. To best fine-tune the calculator, a testing phase was conducted by calculating the carbon footprints of 54 farms located in seven European countries that were characterized by a variety of different practices and products. Wide variation was found in the carbon footprints quantified within each product group. This variation can be explained by different levels of input use, crop yields and other farming practices. It was concluded that the calculator can help EU farmers identify actions that could lead to substantial reductions in the carbon footprints of their farms and products.

Keywords: life cycle assessment, agriculture, greenhouse gas emissions, carbon footprint, environmental footprint

1. Introduction

In the European Union (EU-27 Member States), direct emissions of greenhouse gases (GHG) from agriculture accounted for around 10% of total emissions in 2011 (EEA 2013). According to the European Commission's Roadmap 2050, the target is to reduce agricultural GHG emissions by 42 – 49% relative to 1990 levels by 2050 (EC 2011).

To facilitate reducing farm-level GHG emissions, appropriate and context-specific policy instruments are needed. These measures are to be coupled with supporting tools in order to promote low-carbon farming practices. Many farm-level carbon footprint calculators have been developed to provide support to farmers in identifying the main GHG emission sources along with possible reduction strategies. Examples include the Cool Farm Tool (Hillier et al. 2011), CLA CAML Calculator (CLA 2014), Farm Carbon Calculator (FCC 2014) and Cplan Carbon Calculator (Cplan 2014). As a part of the European Commission project on Low-Carbon Farming, a new farm-level carbon calculator was developed. The Carbon Calculator is suitable for the main farming types in the whole EU, and it also generates farm specific mitigation action recommendations.

During the development process of the Carbon Calculator, the tool was tested on 54 farms around Europe. In this paper the preliminary results of the testing phase are presented and the reasons for differences in the results between different farms are investigated.

2. Methods

2.1. Carbon Calculator

The Carbon Calculator quantifies GHG emissions according to international standards and other technical specifications for life cycle assessment (LCA) and carbon footprinting (GHG Protocol 2011; ISO 14040 2006; ISO 14044 2006; ISO/TS 14067 2013) while also striving for alignment with the European Commission's Environmental Footprint methods (EC 2013) and the EnviFood Protocol (Food SCP RT 2013). The tool is designed to be suitable for the most common farm types in the EU-27.

The main elements of the method underpinning the Carbon Calculator are described below. For a broader description of the method, see Bochu *et al.* (2013). The Carbon Calculator has been developed in Microsoft Excel. Visual Basic for Applications (VBA) was used for creating user forms for data entry. Specific skills for using Microsoft Excel spread sheets are not required to operate the calculator. The tool has been designed to be used by farmers or farmers' advisors. The latest version of the Carbon Calculator, the user handbook and the methodology guide are available for free download at http://mars.jrc.ec.europa.eu/mars/Projects/LC-Farming.

2.2. System boundaries, functional units, allocation rules

The system boundaries for the Carbon Calculator extend from cradle to farm gate. Processing of foods at farm is not included in the system boundaries. The GHG emissions included are CO_2, CH_4, N_2O and hydrofluorocarbons (HFC). GHG emission sources considered are: CO_2 emissions from fuel use and burning of crop residues; CH_4 emissions from ruminant enteric fermentation and manure management; N_2O emissions from soils due to use of organic and synthetic N fertilizers; and HFC emissions from leakage of refrigeration gases. In addition, the upstream emissions generated outside of the farm include emissions from the production and transportation of farm inputs, production of buildings and farm machinery, pumping drinking or irrigation water by collective pumping systems, fuel use by contractors for field operations; and N_2O emissions from NH_3 volatilization and from N leaching and runoff are also incorporated. The user of the Carbon Calculator can choose whether direct land use change (LUC) emissions related to purchased feed are included or not.

Changes in carbon stocks in soils (management practices and land use changes) and in farmland features (natural infrastructure), as well as GHG emissions avoided through the production of renewable energy (whether used on the farm or sold) are quantified but reported separately from other GHG emission results. In addition, the carbon calculator delivers results in terms of direct primary energy use, water use and the nitrogen balance of the farm.

The Carbon Calculator quantifies emissions from the whole farm during a year (or production season). Results are presented on the basis of two different functional units, i.e. (i) the utilized agricultural area (UAA) of the farm (tCO_2-eq/ha UAA) and (ii) a ton of each of up to five main products. In the case of livestock meat production, the functional unit is a ton of carcass live weight. If the farm produces more than five products, the remaining products are allocated to a sixth category named "other products".

In the Carbon Calculator, the production data are entered separately for each crop and livestock type and, in most cases, the inputs can be directly attributed to specific products. In some cases the user has to allocate environmental loads between different products. Regarding the use of fuels in farm machinery (excluding machinery use for crop production), electricity, buildings and other materials used as production inputs (e.g. plastics), the user has to allocate the energy inputs between the products of the farm. Fuel used for field operations is directly attributed to the corresponding crops. The user has to indicate whether each crop is used for feed at the farm or sold out.

In the case of co-products (e.g. milk and meat or eggs and meat) physical allocation based on protein content is used. Meat output is determined based on the weight of the animals sold during the assessment period. The emissions of the whole cattle herd during the assessment period are fully allocated to the products (meat, milk or eggs) delivered by the farm during the assessment period. If the farm does not sell any animals during the assessment period, the carbon footprint of meat is not measured by the Carbon Calculator and all emissions are allocated to the milk sold. Indeed, this is a weakness of the current Calculator that will have to be remedied in its next version.

Two options for the end of life management of exported manure are included: manure is spread on another farmland or treated as residue. If manure is spread on another farm, the farm inventory is credited for the avoided emissions from nitrogen-based fertilizer production calculated as an equal amount of nitrogen from mineral fertilizers. If manure is managed as residue, the emissions from its management are included in the carbon footprint of the farm. In both cases, the emissions from transportation of manure are included.

2.3. Data for emission factors

The IPCC (2006) Tier 2 methodology is used for i) CH_4 emissions from enteric fermentation, manure storage, manure application and manure deposited on pasture land; ii) N_2O emissions from manure storage and application and N fertilizer use; and iii) changes in carbon stocks. The emissions for production and transportation of mineral fertilizers are based on Weiss and Leip (2012), Wood and Cowie (2004), ADEME (2012), GESTIM (2010) and Brentrup and Palliere (2008). For purchased feedstuff the user can choose between two datasets: data based on Weiss and Leip (2012) include LUC emissions whereas data based on ADEME (2012) do not include

LUC emissions. Data from ADEME (2012) is also used for seeds, buildings, machinery, plastics and collective irrigation. Data from the European Life Cycle Data System (ELCD) (2001) are used for electricity and fuels.

2.4. Mitigation and sequestration actions

The Carbon Calculator includes 16 GHG mitigation actions. The actions were selected based on the mitigation potential and practicality of implementation by the farmers. The tool calculates the mitigation costs/savings for six mitigation actions.

2.5. Farm data

Data for the Carbon Calculator were collected from 54 farms in seven European countries, including 20 farms in Spain, 19 in the United Kingdom, 6 in the Netherlands, 4 in Italy, 2 in Germany, 2 in Poland and 1 in Slovenia. The data included 43 conventional, 8 organic, 2 integrated and 1 conservation farm. The study regions were chosen on the basis of their suitability to represent a wide range of environmental zones. The data were collected as a part of a survey that studied the farmers' willingness to use the carbon calculator (except the data from Italy) (Elbersen et al. 2013). A tutorial for the Carbon Calculator was shown to farmers before they were requested to complete a survey regarding their willingness to use the tool. The farmers were subsequently queried as to their willingness to provide farm data for the Carbon Calculator. In total, 170 farmers were approached in eight EU countries (i.e. Denmark, Germany, Spain, Netherlands, Poland, Slovenia, Sweden and United Kingdom) as part of the survey. Of these, 50 farmers provided data for the Carbon Calculator. In addition, data from 4 Italian farms were collected at a later stage.

3. Results

The three main sources of GHG emissions from livestock farms were enteric fermentation, N_2O emissions from soils and manure management. The main emission sources on crop farms included fertilizer production, N_2O emissions from soils and machinery manufacturing (Table 1). The sources with lowest contribution in both farm types were fuels manufacturing and transportation and other inputs (e.g. seeds, pesticides and plastics), and purchased feedstuff for the livestock farms.

The results of the carbon footprint of products show that the median values are close to the reference values found in literature (Table 2), even though the range between minimum and maximum values is wide. The wide range of the results, especially in livestock sectors, is explained by the allocation technique used, which attributes all of the emission of the cattle produced during the assessment period to the livestock product output produced during the assessment period. The high emissions in some crop farms can be explained by high levels of nitrogen fertilizers used on those farms.

Due to insufficient number of samples from organic farms, it was not possible to statistically evaluate the differences between carbon footprints of organic and conventional farms. The results of milk and barley were selected for further analysis as they represent the most common livestock and crop products in the dataset (Table 3). Wilcoxon non-parametric test for sample pairs showed significant difference in the carbon footprint results of milk between the UK and Italy ($W = 6$, $p= 0.022$) and Italy and Spain ($W = 12$, $p = 0.034$) whereas no significant differences were found between the other countries.

In the case of barley production, the statistical difference was tested only between Spain and the UK due to the small sample size in Poland. Significant differences in the carbon footprint results of barley production were not found between Spain and the UK.

Spearman's rank test was used for testing the correlation between nitrogen balance of the farm and carbon footprint results of milk and barley. Significant correlation was not found between nitrogen balance (nitrogen inputs – nitrogen outputs) and carbon footprint of milk production. In the case of barley, statistically significant moderate positive correlation was found between nitrogen balance and carbon footprint ($r = 0.596$, $p = 0.032$).

Table 1. Results of the sources of greenhouse gas emissions from livestock and crop farms.

Emission source	Livestock farms (N=41) % of total emissions	Median ($kgCO_2/ha$)	Variation Coefficient	Crop farms (N=13) % of total emissions	Median ($kgCO_2/ha$)	Variation Coefficient
GHG emissions from direct activities	**87**	**5960**	**1.0**	**43**	**895**	**0.5**
Enteric fermentation	54	3697	1.0	-	-	-
N_2O emissions from soils	18	1256	1.2	32	663	2.1
Manure management	12	797	1.3	-	-	-
Machinery fuel use	4	246	0.9	11	229	0.4
GHG emissions from indirect activities	**13**	**852**	**1.2**	**57**	**1166**	**0.8**
Purchased feedstuff	0.1	4	2.2	-	-	-
Purchased animals	1	60	2.4	-	-	-
Fertilizer production	6	411	0.9	33	682	0.6
Electricity use	3	205	1.4	1	5	2.9
Irrigation	0	6	3.6	-	-	-
Machinery manufacturing	2	109	1.1	20	407	1.4
Farm buildings	0.2	11	2.5	-	-	-
Fuels manufacturing and transportation	0.5	31	0.9	1	28	0.43
Other inputs (seeds, pesticides, plastics)	0.5	17	1.4	2	44	0.79
Total GHG emissions		**6812**	**1.0**		**2061**	**0.55**

Table 2. Carbon footprint results (tCO_2-eq per 1000 kg of crops or 1000 kg of live weight of animals) when land use change related emissions are not included

Product	N	Median	Min	Max	Reference	Source
Barley	15	0.5	0.3	1.4	0.3-0.7	a, b
Wheat	13	0.4	0.2	1.4	0.3-0.8	a, b
Sugar beet	5	0.4	0.1	2.7	0.2	a
Rape seed	7	1.0	0.1	4.0	1.0-1.7	a, b
Milk	27	1.0	0.5	1.8	1.1-1.8	a, b, c
Dairy meat	22	6.2	2.2	14	3.3-4.5	d
Beef	15	18	2.9	77	3.3-47.8	a, b, c, d
Sheep	7	22	13	69	3.5-51.8	a, b, c, d
Pork	5	6.7	2.1	7.7	2.3-6.2	a, b, c, d

a Nielsen et al. (2003)
b Williams et al. (2006)
c Leip et al. (2010)
d Nijdam et al. (2012)

It was found that the results of emission sources for milk and barley production in the inventory correspond closely to the results found in literature for average emissions of milk production in EU and barley production in the UK (Table 4). These results exclude land use change emissions related to purchased feed production. In the case of milk production, the main difference between the median and the reference value was in fertilizer production. This could be explained by the relatively high number of conventional farms in our database that used only manure as fertilizer. The relatively low contribution of capital goods such as buildings and machinery to the overall carbon footprint of milk may be explained by the incomplete submission of data from some farms. Often, it was found out that farmers are seemingly likely to collect little information on capital goods. Comprehensive assessments are difficult without using default data. In the case of barley, the largest difference in the median and reference values was in N_2O emissions from soils. This could be explained by high variation in the nitrogen balances in our dataset.

Table 3. Descriptive statistics of the carbon footprint results of milk and barley production (tCO_2-eq/t).

Country	Number of farms	Min	Max	Average	Median	Standard Deviation	Variation Coefficient
Milk							
Italy	4	1.27	1.55	1.37	1.34	0.13	0.09
Netherlands	4	0.80	1.62	1.07	0.92	0.38	0.36
Spain	3	0.76	0.95	0.84	0.81	0.10	0.12
United Kingdom	14	0.73	1.41	1.03	1.00	0.23	0.22
Barley							
Poland	2	0.24	0.40	0.32	0.32	0.11	0.34
Spain	6	0.43	1.41	0.67	0.49	0.39	0.58
United Kingdom	5	0.42	1.14	0.75	0.73	0.33	0.44

Table 4. Preliminary results of the sources of greenhouse gas emissions from milk and barley production (in kg CO_2-eq/1000 kg milk or barley, N = number of farms).

	Milk (N=25)				**Barley (N=13)**			
Emission source	**Min**	**Max**	**Median**	**Reference value[a]**	**Min**	**Max**	**Median**	**Reference value[b]**
Enteric fermentation	363	842	479	519	-	-	-	-
N_2O emissions from soils	24	491	140	218	114	538	248	178
Manure management	34	358	107	111	-	-	-	-
Machinery fuel use	3.7	112	15	40	27	179	58	51
Purchased feedstuff	0	493	31	50	-	-	-	-
Fertilizer production	0	110	36	136	87	719	170	128
Electricity use	2.8	84	27	50	0	50	6	10
Buildings and machinery	0.2	35	7	80	7	81	19	17
Other inputs (seeds, pesticides, plastics)	0.2	10	3.6	0	0	52	18	12
Total GHG emissions	**733**	**1615**	**985**	**1208**	**259**	**1410**	**485**	**396**

[a] Leip et al. (2010)
[b] Williams et al. (2006)

The most common mitigation actions recommended by the Carbon Calculator were agroforestry, biogas production, and reduction of methane from enteric fermentation (Table 5). The most effective mitigation actions in terms of the median mitigation potential (as % of total farm emissions reduced) included use of no-tillage, improvement of the nitrogen fertilizer balance and biogas production.

The current version of the Carbon Calculator estimates the costs/savings of some of the mitigation actions. In nearly all cases, the Carbon Calculator showed savings gained by implementing the mitigation actions (Table 6). However, these results do not include possible investment costs, but represent only the changes in the input costs.

Table 5. Preliminary results of the mitigation actions recommended to the farms in the dataset (in % of total farm-level emissions reduced).

Mitigation Action	Number of farms	Median	Variation Coefficient
No-tillage	16	9.5	0.87
Adjust N fertilizer balance	16	9.2	1.1
Biogas production	37	8.6	0.77
Agroforestry	45	6.9	1.2
Soils covered all the year	13	5.3	0.64
Reduce methane from enteric fermentation	32	4.8	0.42
Implementation of hedges and other landscape elements	15	4.2	0.60
Introduction of legumes in the rotation	27	3.0	0.82
Change in slurry management system: cover/crust	1	1.9	-
Wood boiler	2	1.8	0.31
Reduce engines fuel consumption (test and eco driving)	10	1.3	0.56
Solar panel on suitable buildings	0	-	-
Introduction of legumes in grasslands	0	-	-
Avoid burning residues	0	-	-
Reduction of electricity consumption of the milking system	0	-	-
Heat water with solar panel	0	-	-

Table 6. Preliminary results of the greenhouse gas mitigation costs/savings

Mitigation action		Mitigation savings (Euro/ha)		
	N	Median	Min	Max
Adjust N fertilizer use	19	93	3.1	1000
Soils covered all the year	14	62	-7.2	1200
Use of wood boiler	5	33	4.0	75
Heat water with solar energy	1	9.6	9.6	9.6
Reduce tractor fuel use	19	8.7	1.7	65

4. Discussion

The preliminary results show that the Carbon Calculator generates carbon footprint results that are close to the reference values found in literature. However, the variation in the results was wide. In the case of livestock meat, the variation can be explained by the allocation method used for allocating emissions to the end product. For instance, in the case of beef, all emissions related to raising the beef cattle are allocated to the meat sold in that year. Assuming a situation when only a relatively small quantity of animals is slaughtered compared to the total number of animals raised in a certain year, the meat produced receives a relatively high apparent carbon footprint. Therefore, the allocation of the impacts to the livestock products does not reflect reality, and thus, caution is needed when the results are compared with other farms or within the same farm between different years.

In the case of crop products, some large carbon footprint preliminary results were explained by high level of nitrogen fertilizer used on those farms. The detailed results of barley production showed that the emissions of fertilizer production varied between 87 and 719 kg CO_2-eq/t barley, and the N_2O emission from soils varied between 114 and 538 kg CO_2-eq/t barley.

The data presented in this paper did not include LUC emissions related to purchased feed production. It has been shown that inclusion of LUC emissions can triple or even quadruple the carbon footprint of livestock products produced in Europe, especially when imported soybean feed is used (Weiss and Leip 2012). We found the same effect in our data when we tested the impact of the choice of purchased feed emission data on the results (data not shown).

The results produced by the Carbon Calculator have to be interpreted with caution. Due to the attributional LCA approach used, the tool does not capture possible impacts associated with indirect effects, such as substitution effect, indirect land use change, income effect, secondary effects, market-clearing price and quantity adjustments. For instance, the emissions of a farm and its products may be reduced by extensification (reduced inputs and reduced yields). However, this may lead in increased production somewhere else, assuming that consumption quantities remain at the same level. Therefore, the impact of the mitigation actions can only be judged when changes in the production patterns and outputs quantities at the farm do not induce market-level changes in production and consumption.

The results presented in this paper showed that mitigation actions recommended by the Carbon Calculator can help farmers to reduce costs while also reducing GHG emissions. This is also supported by literature. For example, MacLeod et al. (2010) showed that, in particular, mitigation actions related to improved nitrogen use efficiency can reduce costs.

In addition to GHG emissions, the Carbon Calculator reports direct energy and water use, and nitrogen balance. To avoid burden shifting from climate change to other environmental impacts, future versions of the tool the scope should be extended to include a broader range of impact categories.

5. Conclusion

This paper showed that the Carbon Calculator generates results that are comparable with results from literature. The Carbon Calculator can also help farmers identify mitigation actions that reduce input costs while decreasing GHG emissions. Due to the methodological choices related to allocation for livestock emissions, comparisons of the results between different farms should be undertaken with caution. In the case of livestock farms, the current version of the tool is best suited for comparing carbon footprint results of one farm within different years if the farm has the same number of animals and animals sold in each year. Further improvements to the tool are required before it can be used for benchmarking purposes.

Other improvement possibilities for the Calculator include harmonization of its underpinning methodology with the European Commission Organizational Environmental Footprint guidelines (EC 2013), including addition of more environmental impact categories. Further harmonization can also be sought against the forthcoming guidelines on environmental assessment of feed and livestock developed in the context of the FAO-led Livestock Environmental Assessment and Performance (LEAP) Partnership. There is also room for adding more mitigation actions, providing uncertainty assessments and providing cost/saving estimates for each mitigation action.

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Environmental impacts of cultured meat: alternative production scenarios

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ABSTRACT

Cultured meat is produced by culturing animal muscle tissue in a laboratory without growing the whole animals. Its development is currently in a research stage. An earlier study showed that cultured meat production could potentially have substantially lower greenhouse gas emissions, land use and water use compared to conventionally produced meat. The aim of this paper is to amend the previous study by considering alternative production scenarios. The impacts of replacing cyanobacteria based nutrient media with plant based media are assessed. This paper includes more specific modelling of a bioreactor suitable for cultured meat production. Further, this study estimates the water footprint of cultured meat based on a method that is compliant with life cycle assessment. The environmental impacts of cultured meat are compared with conventionally produced meat and with plant based protein sources. It is concluded that regardless of the high uncertainty ranges cultured meat has potential to substantially reduce greenhouse gas emissions and land use when compared to conventionally produced meat.

Keywords: in vitro meat, livestock, tissue engineering, carbon footprint, water footprint

1. Introduction

Livestock production contributes 15% of global greenhouse gas (GHG) emissions (Gerber et al. 2013), 33% of the global land use (FAO 2006), and 27% of the global water footprint (Mekonnen & Hoekstra 2011). The consumption of livestock products has been predicted to increase by 70% between 2010 and 2050 (Gerber et al. 2013). Conversion of forests to feed production is one of the main drivers of deforestation and degradation of wildlife habitats.

Tissue engineers have started developing in vitro technologies for producing edible meat. Their aim is to culture animal muscle tissue in a laboratory without growing the whole animals. Currently, cultured meat technology is in a research stage, and the first commercial products are predicted to be available within a decade. An earlier study showed that cultured meat production could potentially have substantially lower GHG emissions, land use and water use compared to conventionally produced meat (Tuomisto & Teixeira de Mattos 2011).

The aim of this paper is to amend the previous study by considering alternative production scenarios. The impacts of replacing cyanobacteria based nutrient media with plant based media are assessed. Cyanobacteria based growth media for tissue culturing is still under development, whereas plant based alternatives are currently available. The previous study had high uncertainty in the estimates of the energy requirements for the bioreactor used for culturing the cells. In this paper, the impacts of a different type of bioreactor, hollow fiber bioreactor, are assessed. Further, this study estimates the water footprint of cultured meat based on a method that is compliant with life cycle assessment (LCA) (Kounina et al. 2013). The environmental impacts of cultured meat are compared with conventionally produced meat and with plant based protein sources.

2. Methods

2.1. Goal of the study

The goal of this study is to estimate the energy use, GHG emissions, land use and water use for industrial scale production of cultured meat. LCA methodology based on ISO14040 and ISO14044 (2006) guidelines is used. The production is assumed to take place is Spain.

The water footprint method was based on recommendation by Kounina et al. (2013) to include only blue water footprint and use country specific water scarcity characterization factors. Both direct and indirect water use is included. Direct water use refers to the direct water inputs used in the process, whereas indirect use refers to the water needed for production of energy sources used in the process.

2.2. Scope definition

The functional unit (FU), towards which all the impacts are allocated, is 1000 kg of cultured meat with dry matter (DM) content of 30% and protein content of 19 % of mass. The cultured meat process described in this paper produces minced-beef type of product as the production technologies for steak type of products are under development. The muscle cells are grown in a bioreactor on a medium composed of a plant-based energy and nutrient source supplemented with the growth factors (proteins that stimulate cellular growth) and vitamins.

The system boundaries cover the major processes from input production up to the factory gate, including production of input materials and fuels, production of the feedstock, and cultivation of muscle cells. The production of growth factors and vitamins are not included in the study as the quantities needed are small (under 0.1% of the DM weight of the media), and therefore, the environmental impacts are assumed to be negligible.

Land use category includes the land requirement for cultivation of feedstock. The indirect land use associated with land use change and the production of energy inputs are not included in the study. The production of the animals that donate the initial cells for cultured meat is not included in the study due to the low number of cells needed.

The GHG emissions are assessed as global warming potential (GWP) by using the IPCC (2006) impact factors for 100-year timescale. The electricity and fuels used are converted to primary energy by using conversion factors that describe the amount of primary fuels required for extraction and supply of fuels. In this study it is assumed that electricity is used for sterilization and muscle cell cultivation. The emission of electricity production are based on an average European electricity generation portfolio.

2.3. Feedstock production

Three alternative feedstock sources of nutrients and energy for muscle cell production are compared: cyanobacteria, wheat and corn. Cyanobacteria are assumed to be cultivated in open artificial ponds. The details of cyanobacteria production are explained in Tuomisto and Teixiera de Mattos (2011). The data for wheat and corn production are based on Williams et al. (2006). After harvesting, the feedstock is sterilized and hydrolyzed in order to break down the cells. It was assumed that 2 kg of wheat or corn was needed for producing 1 kg of cultured meat. This is most likely an overestimate, but more accurate data was not available. The feedstock was assumed to be sterilized by using autoclaving before it was hydrolyzed. The data for blue water consumption or wheat and corn is taken from Mekonnen and Hoekstra (2011). The water scarcity characterization factors were based on data from Pfister et al. (2009), which give index of 0.715 for Spain.

2.4. Bioreactor

To achieve large scale production of cultured meat it is necessary to select the bioreactor configuration that provides a suitable physiological environment. The hollow fiber bioreactor replicates the capillary system found in most tissues. It has the added advantage of having the highest surface area to volume ratio of all bioreactors thereby reducing space and consumables costs.

Heating and pumping energy was balanced against heat energy generated by cell growth. The analysis included a best case and a worst case scenario. In the best case 95% cell viability with maximum theoretical cell yield of 2 x 10^8 cells/mL were assumed and 1 x 10^6 cells as the starting population. In the worst case 80% cell viability with half-maximum theoretical cell yield of 1 x 10^8 cells/mL and 2 x 10^4 cells as the starting population were assumed. The populations under all conditions were assumed to double every two days, and receive the required amount of nutrients and oxygen to achieve these cell densities.

Different sizes of reactors were assumed to be used during the 90 days production period, with increasing the size based on magnitude of volume required. The media was assumed to have the physical properties of water at 37 °C. The bioreactor material was 5mm thick stainless steel and lagging was 25mm thick glass wool. The hollow fibers membranes were made from polylactide. The fiber volume was negligible compared to the media volume and therefore ignored in the heat requirement calculations. The theoretical pumping requirements were calculated and a low efficiency of 0.5 was assumed; selection of the correct pump will improve this efficiency.

The nutrition media was initially heated from 4 °C to 37 °C for the bioreactor volume and for the media that is added during the culture period. The energy required to maintain a temperature of 37 °C is equal to the heat loss to the surroundings, and it was calculated based on the material properties, surface area and temperature differences. It was assumed that the surrounding temperature was 25 °C and that an electrical heating elements were used. The nutrition media was assumed to be changed every three days.

Taking the value of energy released per mole of electrons transferred to be 115kJ, the amount of energy released from the consumption of one mole of oxygen is 460 kJ. Therefore, the heat of reaction, assuming the cell culture is a fully aerobic system, is -460 kJ mol^{-1} of O_2 consumed (Doran 2013). The average oxygen uptake of cells in a culture can be taken as 5.455×10^{-12} mol $cell^{-1}$ day^{-1} (Goudar et al. 2011). This could then be multiplied by the heat of reaction to find the total amount of energy released in a day.

2.5. Comparison with other products

The results of cultured meat production were compared with livestock and vegetable products based on data from literature. The data about GHG emissions and land use of livestock products was from Nijdam et al. (2012), energy use from Williams et al. (2006) and water use from Mekonnen and Hoekstra (2010). When the results were reported in live weights or carcass weights, the conversion factors presented in Table 1 were used to attribute the impacts to edible meat. An economic allocation was used to allocate the impacts between the edible and non-edible parts of the animal. Non-edible part of the animal generally accounts for around 10% of the market value of the animal, and therefore, 90% of the impacts of producing the whole animal were allocated to the edible part.
The results of GHG emissions and land use of cultured meat production were also compared with livestock and vegetable products per unit of protein based on data from Nijdam et al. (2012).

Table 1. Conversion factors used to convert animal live weight or carcass weight to edible meat.

	Beef	**Pork**	**Sheep**	**Poultry**
Carcass weight of live weight	0.53	0.75	0.46	0.7
Edible meat of carcass weight	0.7	0.75	0.75	0.8

3. Results

The major energy input in the cultivation of cultured meat consists of heating energy required to heat the nutrition media and maintain the bioreactor temperature at 37 °C (Table 2).

Table 2. The results of energy requirements for hollow fiber bioreactor in the best and worst case scenarios.

	Best case GJ/FU	**Worst case GJ/FU**
Heating of reactors and media vessels*	8.12	14.60
Mixing media vessels	1.09E-04	1.95E-04
Aeration of media	0.07	0.13
Media pumping	1.93E-06	0.06
Total	**8.22**	**14.78**

*This is a balance of initial heating then maintaining the temperature of the media, and the energy released by the cells.

The results of the full LCA impacts of cultured meat are presented in Table 3. When comparing the feedstock source options, cyanobacteria had the lowest GHG emissions and land use, wheat had the lowest water footprint and corn the lowest primary energy requirement. Bioreactor energy use had the highest contribution to the energy input of cultured meat production. For GHG emissions, land use and water footprint, muscle cultivation and feedstock production had the highest contribution depending on the scenario.

When comparing the results of cultured meat with other livestock products, it was found that energy input requirement was at the same level with beef production, whereas GHG emissions and land use were lower than any of the livestock products (Figure 1). Water footprint was at the same level with poultry.

Table 3. Results of LCA impacts of cultured meat per functional unit (FU) of 1000 kg cultured meat.

	Primary energy GJ/FU	**GHG kg CO_2-eq/FU**	**Land use ha/FU**	**Indirect water use m^3/FU**	**Blue water for cultivation m^3/FU**	**Total water use m^3/FU**
Feedstock						
Cyanobacteria	8.1	611	0.046	6.1	249.5	255.7
Wheat	4.9	1608	0.260	3.7	68.1	71.8
Corn	3.9	1300	0.282	3.0	580.1	583.1
Biomass transportation	0.4	26		0.4		0.4
Sterilization	2.9	144		7.6		7.6
Materials for bioreactor	1.0	108		2.6		2.6
Muscle cell cultivation						
Best case	26.3	1380		57.1	193.1	250.1
Worst case	48.5	2493		57.1	193.1	250.1
Totals						
Cyanobacteria-best case	38.7	2268	0.046	73.8	442.6	516.4
Wheat-best case	35.5	3266	0.260	71.4	261.1	332.5
Corn-best case	34.5	2958	0.282	70.6	773.2	843.8
Cyanobacteria-worst case	60.9	3381	0.046	73.8	442.6	516.4
Wheat-worst case	57.7	4379	0.260	71.4	261.1	332.5
Corn-worst case	56.7	4071	0.282	70.6	773.2	843.8

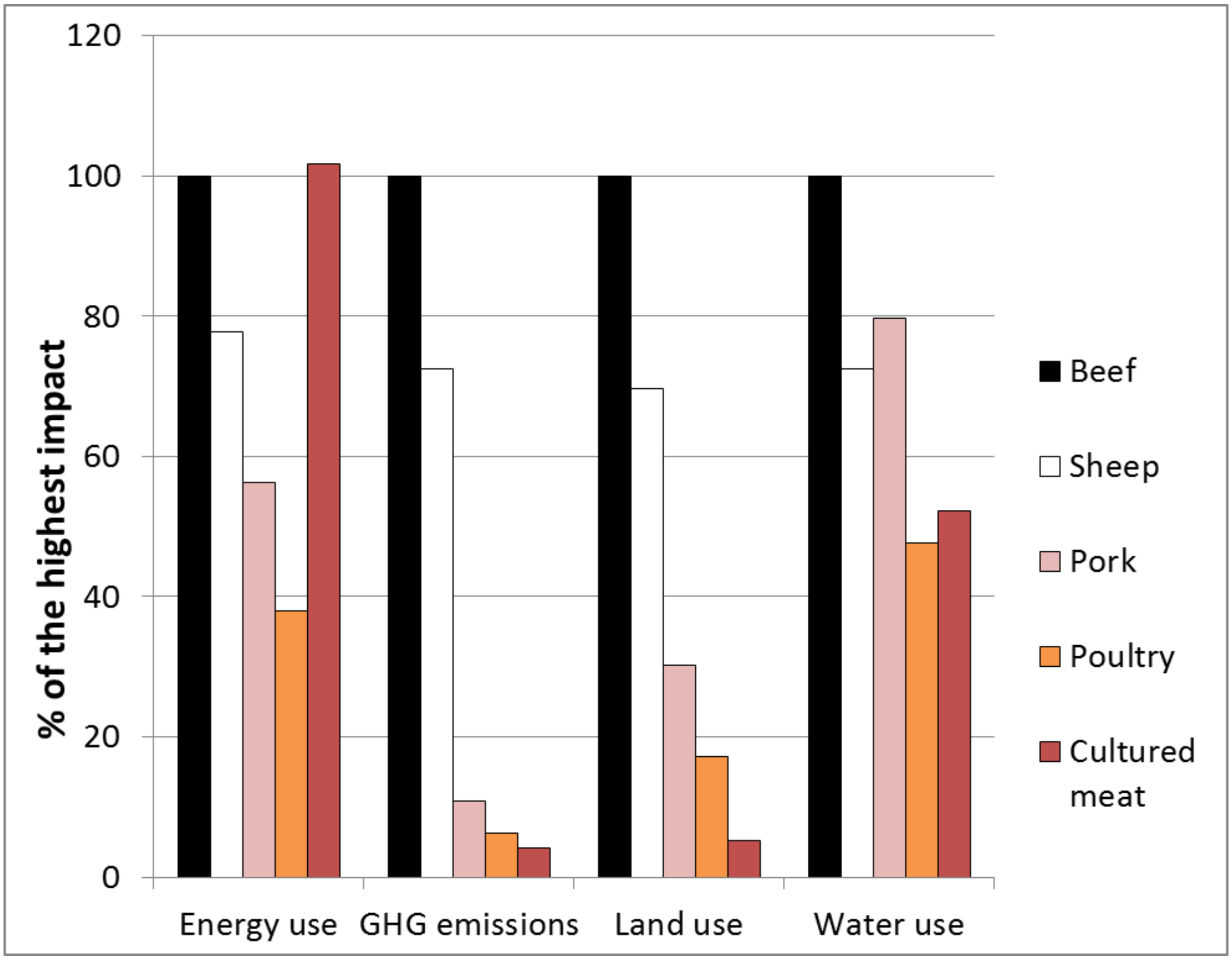

Figure 1. Comparison of environmental impacts of cultured meat with European livestock meat.

Figure 2 and 3 show the GHG emissions and land use of cultured meat compared with animal and plant based protein sources. The GHG emissions of cultured meat are at the same level with plant based protein and animal protein with the lowest carbon footprint. In terms of land use, cultured meat has the lowest land use requirements.

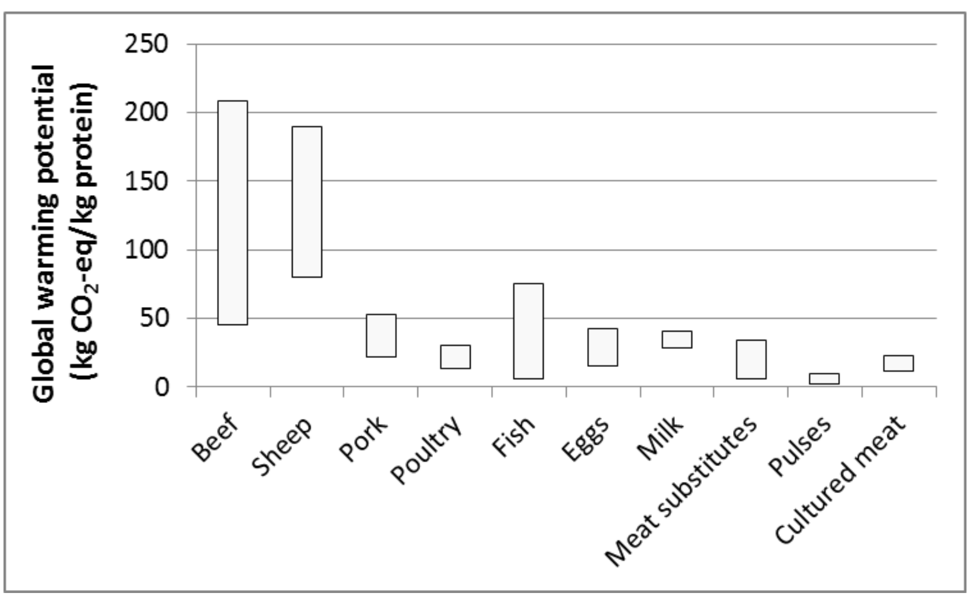

Figure 2. Comparison of GHG emissions of cultured meat with animal and plant based protein sources.

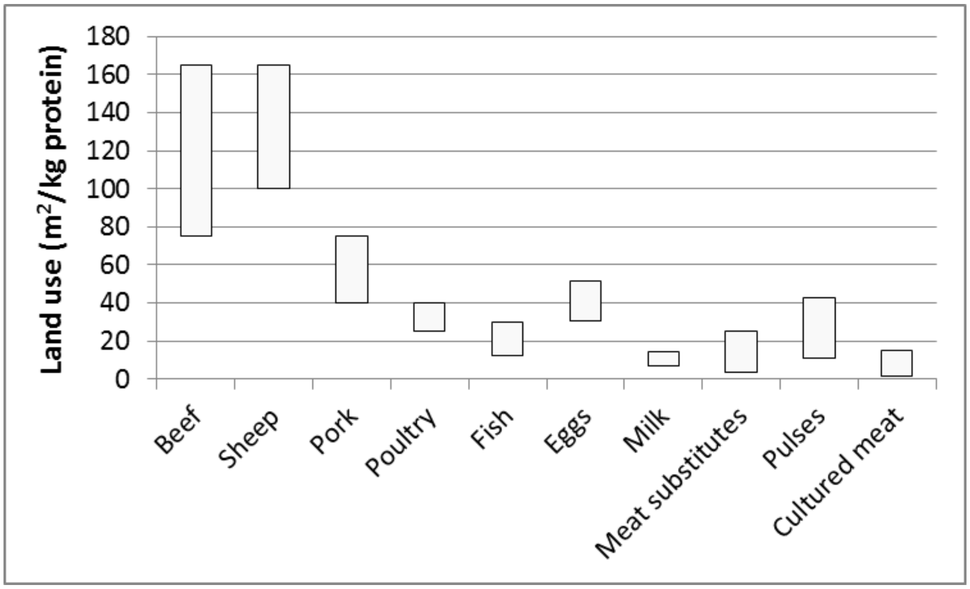

Figure 3. Comparison of land use of cultured meat with animal and plant based protein sources.

4. Discussion

As cultured meat production is still in research stage, the results presented in this paper have high uncertainty. The main uncertainty is related to the design of the bioreactor and composition of the nutrition media. Both have major impact on the environmental contribution of cultured meat production. The results of the hollow fiber bioreactor use are only preliminary hypothetical calculations. The energy requirement may be reduced by modifying the process, for example, by using heat exchangers. More research is required for developing suitable bioreactors for large scale cultured meat production.

Further research is also required for development of plant-based nutrition media for cell culturing. In this paper, only the impacts of producing the main ingredients of the nutrition media were included. More detailed assessment of impacts of producing the media should be performed once more information about a suitable plant-based media for cell culturing becomes available.

When comparing the current study with the results of Tuomisto and Teixeira de Mattos (2011), the main differences can be found in the energy input and water footprint results. The reason for higher energy input of cultured meat in this study is explained by more accurate modelling of the bioreactor and inclusion of bioreactor heating requirement.

The relative difference in the water footprint results is explained by updated methodology that includes only blue water footprint. Livestock production has high green water footprint, but relatively low blue water footprint.

5. Conclusion

We conclude that the uncertainties in the environmental impacts of cultured meat remain high. The alternative production scenarios compared in this paper help to better understand the sources of these uncertainty ranges. Regardless of the high uncertainty the results show that cultured meat could have substantially lower GHG emissions and land use when compared to conventionally produced meat. It is also important to take into account the consequential impacts of reduced land use requirements. If a large proportion of meat was produced by using cultured meat technology, land would be released from meat production for other uses. This land could potentially be used for provision of ecosystem services (e.g. for forests or natural conservation areas), which would increase the total environmental benefits of cultured meat production. More research on development of cultured meat technology is needed before more reliable estimates of the environmental impacts can be provided.

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Just eating healthier is not enough: studying the environmental impact of different diet scenarios for the Netherlands by Linear Programming

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ABSTRACT

Eating healthier or vegetarian and vegan diets are suggested options to reduce the environmental impact of the current diet. In this paper we investigate different scenarios and assess the reduction of environmental impact after restoring the nutritional adequacy by replacements. We used Linear Programming to find solutions that are as close as possible to the current diet, first without any food groups' constraints and later by imposing constraints on meat, fish, dairy and eggs. Finally, we use a similar technique to search for the closest diet that achieves the same environmental reduction as the most restricted option (no meat, fish, dairy or eggs), without restrictions on products. We show that it is possible to find less restrictive solutions than vegetarian or vegan diets that satisfy all nutritional requirements and have less environmental impact than the current one. Most important, these are closer to the current diet.

Keywords: Linear Programming; Food Policy and Nutrition; Environmental Impact; Greenhouse gas emissions; Food patterns

1. Introduction

Global food production faces a big sustainability challenge, which comprises many aspects. Global agriculture is the main contributor to biodiversity loss, water resource depletion and soil degradation due to the extent of land occupation and current farming practices (Searchinger et al. 2013). Searchinger et al. (2013) states also that the production of crops and animal products today releases roughly 13 percent of global greenhouse gas emissions (GHGe); the conversion of forests, savannas, and peat lands to agriculture roughly accounts for an additional 11 percent of global GHGe and that the ongoing expansion of cropland and pastures is the primary source of ecosystem degradation and biodiversity loss. Naturally, these estimates are uncertain and better monitoring of agricultural emission is needed (Tubiello et al. 2014). However there is broad consensus that in order to decrease the demand for natural resources and reduce GHGe food consumption and production have to become more sustainable, especially because the world's population is still increasing rapidly. The Food and Agricultural Organization (FAO) estimates that by 2050 agricultural production has to be increased by 70 percent (FAO 2011).

The central question is how to decrease the impact of the food system. There are two complementary routes: improvements on the supply side and changes in the demand of foods. Sustainable intensification of crop production (FAO 2010) and implementing technologies and practices to reduce emissions from livestock production (Gerber et al. 2013) have huge potential, as can be illustrated by the differences in GHGe from dairy worldwide (Hagemann et al. 2012). Changing food patterns and reducing food spoilage at home are some of the more direct options at the demand side. This paper focusses on food patterns with a lower environmental impact.

Several authors suggested that, for example, making healthier choices or adopting to Mediterranean type diets could reduce the impact (Duchin 2008; Carlsson-kanyama and Gonzalez 2009; van Dooren et al. 2014). This was contradicted by Vieux *et al* (2013) who compared self-selected diets from a French dietary survey on GHGe. They also pointed out that this approach does not take into account the effort that consumers have to make to adopt these dietary changes. An elegant way to design alternative options is to apply Linear Programming (LP) to minimize the amount of necessary changes in the current diet (Maillot et al. 2010). At the same time the inclusion of nutritional restrictions in the LP model safeguards an adequate intake of nutrients and energy.

In this paper we study the environmental impact of a few scenarios. We start with the current average Dutch diet of women between 31 and 50 years of age and search for the closest healthy diet, which satisfies all nutritional requirements. This will show that just eating healthier is not a guarantee of a more environmentally friendly diet. We continue by analyzing three predefined dietary interventions in which different types of animal-based products are eliminated from the diet. In particular we look at two common vegetarian-type diets and a vegan diet that also excludes dairy and eggs. Finally, we compare the necessary changes in these scenarios with an optimal solution calculated by LP after imposing only a restriction on the total environmental impact, using the same target as can be reached maximally with the scenarios with product restrictions. Based on these analyses we obtain a good impression on how to find a food pattern which is not only healthy, but also low on environmental impact. Moreover, this diet pattern remains as close as possible to the current average situation.

2. Methods

2.1. Reference Profile and Nutritional requirements

As a case we have chosen a non-active woman between 31 and 50 years of age. This group was chosen because it has a relatively high requirement for iron, which can become critical if meat, an important source of iron, is removed from diets (Macdiarmid et al. 2012). Nutritional requirements were defined in close cooperation with the Netherlands Nutrition Centre (Voedingscentrum, The Hague) and are a compilation of the Dutch Food-Based Dietary Guidelines (FBDG) (Voedingscentrum 2011), national (Gezondheidsraad 2001; Gezondheidsraad 2006) and international recommendations (WHO et al. 2007) for energy and nutrients. To promote variation of diet, constraints were set on the maximum amount of each product in a weekly diet.

2.2. Current average Dutch diet

A simplified weekly current average Dutch diet is modelled with 207 products out of 1599 consumed by the survey population (n=3819) of the Dutch Consumption Panel (Rossum et al. 2011). This selection consists of the items contributing most to the total intake of the survey population (about 80% by weight). To compensate for the eliminated products, we scaled the remaining ones in such a way that the total amount within each product group was equal to the original average. The nutrient value of the observed average diet and our current diet was very similar[1].

2.3. Nutritional and Environmental data

Food composition data, including macro- and micronutrient content, was obtained from the Dutch Food composition table (RIVM 2011) while data on essential amino acids was collected from the United States Department of Agriculture Database (USDA 2012). Representative data on Greenhouse Gas Emissions (GHGe), Fossil Energy Use (FEU) and Land Occupation (LO) of all foods was collected from various sources and modelled from cradle to grave, including retail phase, the consumer phase and end of life of packaging materials (Kramer et al. 2013). Corrections were made for un-edible parts, raw to cooked ratios (RIVM 2012) and avoidable waste (Van Westerhoven and Steenhuizen 2010).

The individual environmental indicators were used to calculate a weighted score, based on the ReCiPe method (Goedkoop et al. 2013), which we call *p*ReCiPe. By combining the characterization, normalization and weighing factors of these three indicators we reach the following expression:

*p*ReCiPe=0.0459*GHGe+0.0025*FEU+0.0439*LO
GHGe = kg CO2-equivalents/kg
FEU = MegaJoules/kg
LO = m2*year/kg

We omitted other contributing midpoints, due to a lack of reliable and consistent inventory data for all products in scope and the fact that these 3 indicators are the most important contributors to the ReCiPe single score (Sevenster et al. 2010).

[1] Comparative tables are available upon request.

2.4. Linear programming and a metric for changes (penalty score)

Optimization of diets was performed with Optimeal®, a commercial Linear Programming (LP) tool developed by Blonk Consultants. The algorithm is described in detail elsewhere (Kramer et al. 2013). In the LP model amounts and deviations are expressed per serving (Voedingscentrum 2013). The goal of the LP is to minimize changes such that the solution diet satisfies all nutritional and, optionally, environmental constraints.

The metric for changes is operationalized by a penalty score which, in turn, reflects the popularity of foods. Any deviation from the current diet contributes to the penalty score, but the penalty contribution of each serving change is food and directional dependent. More specifically, the amount of serving changes in a given food is multiplied by a normalization of the total quantity (grams) consumed of that food during the dietary survey (RIVM 2012). In that way, the more popular a food is, the lower its penalty for increases is and the higher its penalty for decreases is. Conversely, the less popular a food is, the higher its penalty for increases is and the lower its penalty for decreases is. The penalty score can be interpreted as a measure of distance between diets.

The reasoning behind this modelling is the principle that diets which are more similar to the current one are more likely to be accepted by the majority of the population than more extreme diets. Also removal of popular products or introduction of unpopular products is not likely to be easily taken into practice. By comparing the penalty scores of different diets it is possible to measure how similar they are to the current diet.

2.5. Dietary Scenarios

We study 6 dietary scenarios. We start by the current average Dutch diet (*Current Diet*), modelled as described above and use it as the starting point of all optimizations. The other 5 dietary scenario are optimized diets (minimal penalty score) such that the nutritional constraints are always met, thus healthy diets. The difference among the diets is the (additional) constraints imposed. The first scenario, *Closest healthy* diet, does not add any product or environmental constraints. Next we study 3 commonly discussed diets in the context of defining sustainable diets (van Dooren et al. 2014): a *Vegetarian M* diet, which excludes meat products; a *Vegetarian MF* diet, which excludes meat and fish and a *Vegan*[2] diet, which excludes meat, fish, dairy and egg products. Finally we look at a diet with 30% less environmental impact (*30% less*), which instead of imposing constraints on food groups includes only a constraint on the environmental impact. We choose this target, because it was the largest reduction achieved by the other pre-selected diets, which was in fact the vegan diet.

3. Results

The current diet in the Netherlands does not meet all dietary guidelines (Rossum et al. 2011), as in many other countries (Mensink et al. 2013) (see numbers in bold in Table 1). For instance, it is too low in dietary fiber and several vitamins and minerals, whereas the intake of saturated fat is too high.

To obtain a healthier diet that is as close as possible to the current one, more products have to be added than deleted (Table 2). The *p*ReCiPe score remains the same. Both GHGe and LO decrease, whereas FEU increases. This is mainly due to a increase in the amount of fish[3], and fishing is energy intensive. Furthermore, the *Closest healthier* diet contains more fruit, vegetables and legumes. The latter supplies both fiber and vitamin B1, which were below the recommended intake in the *Current* diet. To reduce the intake of saturated fatty acids (SAFA) several types of sausage and cheese are reduced.

[2] In the two dietary scenarios which exclude fish, it is not possible to meet the dietary guidelines for DHA+EPA. We had, thus, to ignore this nutritional constraint is these cases.

[3] The involved species, such as herring and farmed salmon, are currently not overfished.

Table 1. Constraints (Lower and Upper) in the Linear Programming model and properties of the current diet (per day):

Property	Lower	Upper	Current[1]
Energy (kcal)	1890	2110	1993
Protein (g)	50	125	84
Fat (g)	44	89	79
Saturated FA[2] (g)	-	22	**28**
Polyunsat. FA[2] (g)	-	27	15
Linoleic acid (g)	4	-	13
α-linolenic acid (g)	2.2	-	**2**
Trans FA[2] (g)	-	2.2	1
Cholesterol (mg)	-	300	191
Carbohydrates (g)	200	350	216
Dietary fiber (g)	30	-	**19**
Water (g)	2300	3800	3101
Alcohol (g)	-	10.0	4
DHA+EPA[3] (mg)	450	1000	**172**
Retinol act eq (µg)	700	3000	**668**
Vit B1 (mg)	1.1	-	**0.8**
Vit B2 (mg)	1.1	-	1.3
Niacin (mg)	13	-	18
Vit B6 (mg)	1.5	25	**1.4**
Folate equivalents (µg)	300	1000	**231**
Vit B12 (µg)	2.8	-	4.4
Vit C (mg)	75	-	80
Vit D (µg)	3.3	100	**2.8**
Vit E (mg)	8.0	300	12
Vit K (µg)	90	-	115
Calcium (mg)	1000	2500	1100
Phosphorous (mg)	600	3000	1550
Iron (mg)	15.0	25.0	**10.0**
Sodium (mg)	-	2400	2388
Potassium (mg)	3100	-	3330
Magnesium (mg)	280	530	334
Zinc (mg)	7.0	25	11
Selenium (µg)	50	300	**46**
Copper (mg)	0.9	5.0	1.1
Iodine (µg)	150	600	169
Tryptophan (g)	0.3	-	0.9
Threonine (g)	1.1	-	2.9
Isoleucine (g)	1.5	-	3.6
Leucine (g)	3.0	-	6.5
Lysine (g)	2.3	-	5.9
Methionine (g)	0.8	-	1.9
Cystine (g)	0.3	-	1.0
Valine (g)	2.0	-	4.3
Histidine (g)	0.8	-	2.4
Vegetables (g)	200	500	**139**
Fruit (g)	200	500	**124**

[1] *Bold numbers indicate values outside the constraints*
[2] *FA = fatty acids*
[3] *EPA (Eicosapentaenoic acids), DHA (Docosahexaenoic acid): both omega-3 fish fatty acids*

If we subsequently add constraints on meat and optimize to find the closest adequate solution (*Vegetarian M*), both LO and GHGe are lower than the initial value of the current diet. The *p*ReCiPe is reduced by 17%. Omitting meat from the current diet (*Vegetarian M*) forces more changes, because it is an important source of several nutrients that are already below the lower limit in the current diet: e.g. B1, B6, selenium and copper. To replace these nutrients from meat substantial amounts of cheese, fish, legumes and meat replacers are added to the diet. This explains the relatively modest reduction of impact in this scenario. Because of their high sodium content soups (goodies) are removed to make space for cheese. When we subsequently restrict fish (*Vegetarian MF*) and dairy and eggs (*Vegan*) the environmental impact decreases. To replace the essential nutrients from fish (e.g. vitamin B12 and D), a fortified soy drink, eggs and nuts are added to the *Vegetarian MF* diet. The absence of dairy and eggs in the *Vegan* diet has an additional impact on the nutrient intake. Alternative sources of vitamin B12, calcium, essential amino acids and many other nutrients have to be found in compensation. For instance, brown rice as a source of Se and niacin, vegetables as a source of vitamin K and retinol equivalents and more fortified soy drink for vitamins B2, B12 and D.

The reduction in *p*ReCiPe obtained with the *Vegan* diet is 0.13 points or 30%. The same reduction can be realized without forced restrictions on animal-based product groups. Interestingly, this "*30% less*" diet shows lesser shifts in amounts per product group than the *Vegan* diet in comparison with the *Current* diet. Savings are not only found by reduction of meat, beverages and goodies[4] but also by making more sustainable choices within categories[5]. The contribution of meat in total savings is more than 60% in comparison with the *Closest healthy* diet. Within the meat category both beef and chicken are reduced, and mainly pork products remain. This is an interesting finding, because chicken has a lower environmental impact per unit. Most likely, the balance between nutritive value and environmental impact is less favorable in chicken. More specifically pork contains more of several critical nutrients like Fe, Cu, Se, B2 and B12 (RIVM 2011). Within the beverages group beer and wine are reduced. In the group goodies soups are reduced but some apple syrup is added. The latter is added due to the regular fortification with Fe, a nutrient likely to become critical when meat is reduced. Dairy (liquid and cheese) even increases, when expressed in milk equivalents (1 gram of cheese is equivalent to 8.4 grams of milk (Kramer et al. 2013).

Within the fruit category apple sauce is replaced by some banana and melon. Other replacements are cheese for buttermilk (dairy) and herring for salmon and mussels (fish).

The amount of changes relative to the current diet are reflected by the penalty score (Table 2). Quite some changes are necessary to make the diet just healthier (*Closest healthy*). The more animal-based products are restricted, the higher the penalty score is after restoring the nutritive value. These restrictive strategies have a higher penalty score than the optimal solution when the upper limit for *p*ReCiPe is set at 0.29 points.

To illustrate how complex the implementation of these dietary scenarios can be and how many other options are overlooked, we plotted (Figure 1) the environmental score (*p*Recipe) of the 6 studied dietary scenarios against their penalty score. The plot also contains a possibilities frontier line which is calculated as the minimum penalty score such that a given environmental score level is achieved.

[4] Non-staple foods, with a low contribution to the nutrient intake (Voedingscentrum 2011). These include, e.g., sugar, cookies, snacks, sauces, jam, candy, etc.
[5] Due to space constraints detailed tables with the breakdown were not included. These are available upon request.

Table 2. Quantities of foods and environmental indicator in diets. Amounts per day.

	Current	Closest healthy	Vegetarian M	Vegetarian MF	Vegan	30% less
Food groups						
Potatoes, pasta, rice, etc. (g)	136	136	136	136	217	136
Bread (g)	145	145	145	145	163	145
Vegetables (g)	139	200	200	200	422	200
Fruit (g)	124	200	200	200	200	200
Dairy (liquid) (g)	334	313	334	312	0	280
Cheese (g)	38	9	22	21	0	15
Meat (g)	91	83	0	0	0	28
Fish (g)	18	41	80	0	0	40
Egg (g)	11	11	11	50	0	11
Oils, fats, fat spreads (g)	27	27	27	25	22	27
Nuts, seeds (g)	7	11	12	38	2	12
Water (g)	794	794	794	794	794	794
Beverages (g)	1451	1483	1451	1565	1828	1404
Legumes (g)	4	42	92	107	120	70
Meat replacers (g)	0	0	86	50	0	0
Goodies (g)	208	300	142	129	167	259
Environmental indicators						
GHGe (kg CO2eq)	4	3.6	3.1	2.7	2.4	2.5
Fossil Energy (MJ)	32	36	36	30	31	28
Land Occupation (m2*a)	4	3.8	2.7	3.1	2.4	2.4
*p*ReCiPe (Pt)	0.42	0.42	0.35	0.33	0.29	0.29
Penalty Score	0	63.01	100.76	138.20	365.5	89.44

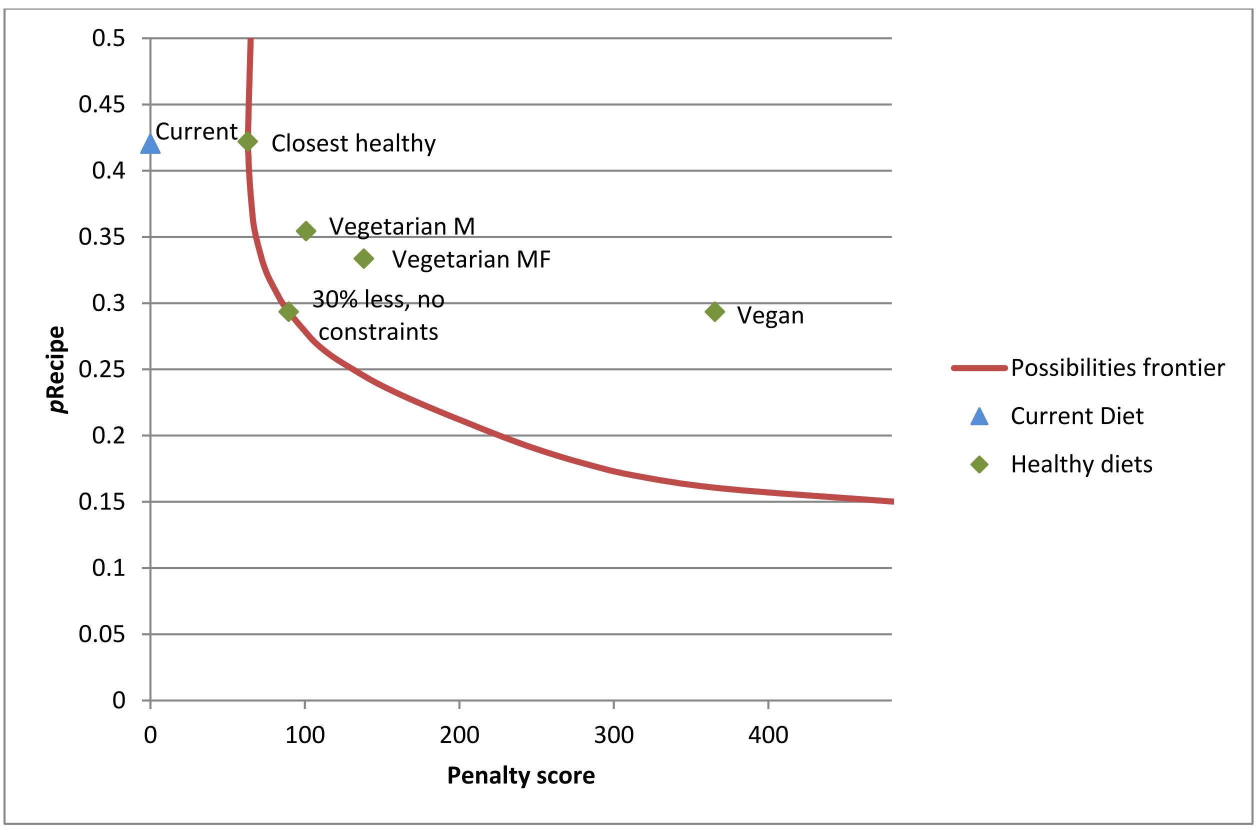

Figure 1. Penalty scores and environmental impact of the several dietary scenarios

In the figure, the closer a diet is to the frontier line, the more similar it is to the current a diet, while being healthy. The *Closest healthy* and the *30% less* are, by construction, on the frontier. Notice how much further the *Vegetarian* and *Vegan* diets are from the frontier. This does not indicate *per se* that a *Vegetarian* or *Vegan* options are not valid, but it does indicate that, if the goal is simply to reduce the environmental impact, there are many other options possible to reach the same environmental goal with less changes. These overlooked options still include animal-based products and are more similar to the current situation and, therefore, more likely to be accepted by a general population. Notice this diet resembles a so-called *Flexitarian* diet, which promotes the reduction of animal-based products, as a pre-defined intervention. This diet design is here, however, an outcome of a structured search procedure. Moreover, we selected the closest diet to the current which satisfies all nutritional requirements and achieves a 30% reduction of the environmental impact, the same level as the healthy *Vegan* scenario.

4. Discussion

4.1. Main observations

For the first time we compare different dietary scenarios using a metric for the distance between diets, taking popularity and environmental indicators into account. This improves on previous literature (Maillot et al. 2010). Our results indicate that just eating healthier is not necessarily beneficial for the environment. This supports the conclusions by Vieux *et al* (2013) who compared diet quality and GHGe from actual diets in France. The distance to the average current diet is larger when certain changes are enforced instead of left open. With relatively minor changes the environmental impact of a healthy diet (*Closest healthy*) can be improved with 30%. The optimal solution still contains 30% of the amount of meat quantity in the *Closest healthy* diet, whereas amounts of dairy (liquid and cheese), fish and egg almost remain constant. The reduced meat consumption is responsible for 60% of the reduction in environmental impact. Additional savings are realized in other categories like beverages and goodies. Our results also show that replacement scenarios of animal-based products should not be focused only on finding alternative sources of protein (Westhoek et al. 2011). To meet guidelines other nutrients like Fe, Ca, Se and vitamins B12 have to be replaced as well and might even be are more critical. In a comparison between the *Current* diet and the closest with 30% impact, reason for changes cannot be distinguished. Therefore, it is more meaningful to compare the *Closest healthy* diet with the closest with *30% less* impact. In this comparison all changes are induced by the restriction on the environmental indicator *p*ReCiPe. It shows that meat and beverages are still reduced, but also goodies (other than staple foods). Dairy (liquid and cheese) on the other hand even increases, when expressed in milk equivalents. Also legumes increase.

4.2. Validity and limitations

With the 207 products in the model we cover approximately 80% of the intake by weight. The other 20% represent 1392 products. It is possible that some of these products have a favorable nutritional and or environmental profile so that they would have been selected in the optimization. The quality and representativeness of included LCA data varies. For important product groups like meat, dairy, bread, legumes and vegetables we relied on data from a reviewed comparative LCA (Kramer et al. 2013). These data were obtained by applying a consistent LCA method regarding for instance allocation and system boundaries. Still, the quality of the LCI data in these studies varies. We are aware that data quality and variability could influence the outcome of the study..

Limitations of our environmental indicator are that it does not account for overfishing and makes no distinction between types of land use. We verified if this had an undesirable influence on the outcome, which was not the case. Overfished species were not increased. With regard to land use type, when this was taken into account a similar trend was found as in overall land occupation.

There is also variability in the food composition data we used but the present values represent the best estimate for products consumed in the Netherlands (Westenbrink and Jansen-van der Vliet 2013).

A limitation of the present study is that we only studied one group (women 31-50). We did not verify how different the outcome would be for men or other age groups. Differences could occur due to different nutritional requirements and due to different average diets. Another limitation is the lack of a good indicator for marine resource depletion. By ignoring this environmental impact a bias towards higher consumption of fish is introduced.

Finally, complex system inter-relations are not taken into account. For example, beef from the dairy production system and dairy products are not linked in the present model: they can be decreased or increased independently and different production systems are also not taken into account in this study. In the current system approximately 17 grams of beef is produced with every liter of milk. The diet with *30% less* environmental impact contains a quantity of beef that is higher than the amount associated with the suggested amount of dairy in the diet. The vegetarian diets, on the other hand, create a surplus of beef from dairy production. To incorporate these concerns an extended model, which is beyond the scope of this paper, would be necessary.

4.3. Sensitivity

The outcome of LP is sensitive to choices in restrictions and formulation of the objective function. Replacing the penalty score by a simple Euclidian Distance or the count of number of product changed, however, does not alter our qualitative analysis. The sensitivity to restrictions can be illustrated by the following observation. Some nutrients are almost exclusively delivered by one product group. In those cases the recommended amount of such a product group is very dependent on the level of one specific guideline. One example is the guideline for omega-3 fish fatty acids (DHA+EPA) in our model. Solutions without fish are therefore impossible and compared to the current diet, fish consumption should even increase to obtain a healthier diet. Wilson *et al* (2013) on the other hand, found optimized diets without fish because they did not include such a guideline.

4.4. General outcome

The average current diet in the Netherlands does not meet all dietary guidelines which are present as constraints in the Linear Programming (LP) model. The changes necessary to make it healthier, taking the popularity of products into account, does not change the environmental impact. Enforcing pre-defined dietary interventions, such as vegetarian or vegan diets has benefits for the environment, but results in an inadequate intake of omega-3 fish fatty acids when fish is left out of the diet. There is strong evidence that these fatty acids have a protective effect against fatal coronary heart diseases.

The choice for vegetarian or vegan diets in scenario studies can be argued to be arbitrary and ignore a whole set of possible diets which also satisfy nutritional constraints and yield a comparable or even lower environmental impact. The environmental improvement goal of 30% can be met by less drastic changes to the diet, if penalty scores of the different scenarios are compared. Another advantage is that substitutions are chosen based on objective criteria: a balance between nutritive value, popularity and environmental impact. Vegetarian diets have a systematic flaw because they contain dairy, but not the meat produced in dairy systems. Also solutions without fish have flaw, because they do not meet a guideline for omega-3 fish fatty acids. Our data shows that the reductions contributing most to the decrease of impact relative to the current diet are indeed in meat and cheese, and certain beverages like beer, wine and soft drinks, but they are all still part of the diet. On the other hand, legumes, fruit and vegetables are the most important additions to the current diet necessary to restore or improve the dietary quality.

The results we obtained when the constraints on groups were removed are quite similar to those obtained by MacDiarmid *et al* (2012). They also saw decreases in animal-based products and increases in plant-based foods. Because they ignored beverages, they missed the potential saving in this category. Because we observed similar trends we can confirm the validity of other studies that only used GHGe (Risku-Norja et al. 2009; Tukker et al. 2009; Stehfest et al. 2009; Vieux et al. 2012; Macdiarmid et al. 2012; Wilson et al. 2013). Only a few studies have addressed other impacts or used a single environmental indicator (Gerbens-Leenes and Nonhebel 2002; Tukker et al. 2009; Temme et al. 2013).

5. Conclusion

Within the Dutch context, eating according to nutrient requirements does not necessarily have a lower environmental impact. For women in the group of 31-50 we found that the closest healthy diet relative to the current diet in the Netherlands has the same environmental impact. It is possible however to find many other diets with reduced environmental impact which are healthier than the current situation, still includes meat, fish and dairy and eggs by using linear optimization. The method we propose is an innovative and systematic way of finding improved diets which are close to the current average diet satisfying all nutritional and additional environmental constraints.

Providing a diet that meets dietary requirements is a prerequisite for (sustainable) diets. Simply omitting animal-based products from the diet is not a valid option: other products have to be added to guarantee nutritional adequacy. In that respect, protein should not be the only focus, as an adequate intake of other nutrients supplied by animal-based products are also at risk.

The scenarios presented in this paper illustrate that omitting animal-based products from the diet require substitutions to meet dietary requirements. These, in turn, reduce the positive environmental impact change. For the general populations these scenarios seem less attainable than a scenario in which the desired reduction in impact is added as a binding constraint in the LP model. In that case reductions in meat (beef, chicken) and beverages (beer, wine) are making the most important contributions. Eating of legumes and (sustainable) fatty fish should be promoted. On the other hand, consumption of goodies should be discouraged, both for health and for the environment.

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sigAGROAsesor: A software platform application to extend the use of sustainability indicators into agricultural systems

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ABSTRACT

Food systems will have to provide increasing amounts of products at lower environmental costs. Life Cycle Analysis (LCA) is a methodology that can be used in agricultural systems to quantify input flows, materials and energy, as well as processes needed to obtain a food product. Selection of the appropriate impact categories and use of the corresponding indicators is of upmost importance to promote agricultural sustainability. A key aspect to guarantee the success of the concept and use of environmental impact thinking in agriculture is the involvement of farmers in the calculation of sustainability indicators. The objective of this work is to develop an online GIS-based platform for extensive crops with four decision-support tools (DST): crop varieties, fertilization, irrigation, and risk of plant disease appearance, at the same time that agricultural sustainability indicators can be calculated, in a plot basis. Selected indicators are: a) Carbon footprint, b) Water footprint, and c) Pesticide application pressure, ecotoxicity and human toxicity. Trade-offs among indicators are taken into account by using a per product unit or per area approach to balance out the caused impacts. In this way, we expect that main impacts of agricultural systems will be controlled at the same time that increasing crop productivity is achieved.

Keywords: LCA, GIS, Carbon footprint, Water Footprint, Pesticide toxicity

1. Introduction

At European level food (particularly, meat and dairy products) are among the sectors causing the majority of environmental impacts related to final consumption expenditure (Tukker et al., 2006). The manufacture of fertilizers and pesticides, farm operations, processing and transport of farm products, packaging, refrigeration, cooking, and end of life disposal options depend on natural resources and fossil energy. Most of investigations related to environmental aspects associated with agriculture focus on specific items, such as ammonia volatilization, nitrous oxide emissions, nitrate leaching, phosphorous fixation in soils as well as P losses, etc. However, in order to evaluate and compare the entire environmental burden related to agricultural production systems, it is necessary to consider all environmental impacts at the same time (Brentrup et al., 2001).

Sustainability comprises three pillars: economic, environment and social, and many indicators have been proposed to assess the state of the sustainability of agricultural production systems. Life Cycle Analysis (LCA) is a methodology used to measure the environmental impact of a product, process or system through its entire life cycle. In addition to identifying the impacts and potential improvement options of a product, LCA can aid in the selection of relevant indicators of environmental performance (ISO 14040:2006).

Looking at only one or a few indicators, important aspects of sustainability can be missed, so it is needed to choose a set of indicators that fit properly the system we are evaluating. Several schemes have been proposed in order to account for a balanced set of environmental indicators in agriculture; for example, under the GLOBAL 2000 scheme (Wildenberg, 2012), five field level-based indicators (N-balance, P-balance, humus-balance, pesticide use and energy intensity) and five indicators based on "material input per service unit" (carbon-footprint, biotic and abiotic material input, water input and area used) were chosen. With a focus on the on-farm activities, Agbalance™ Life Cycle Analysis combines environmental, social and economic sustainability indicators, calculates aggregated indices for the three types of indicators, and provides a single overall sustainability index (Schoeneboom et al., 2012).

In recent decades, LCA has been increasingly used for the assessment of agricultural impacts (Anielski Management Inc, 2010; Point, 2008; Renouf and Fujita-Dimas, 2013) due to its utility to inform strategic environmental programs, monitor progress, and most importantly, lead to a minimization of environmental burdens resulting from the provision and use of products and services (Guinée et al., 2002).

The life cycle inventory (LCI) is usually the most time-consuming and complicated stage of an LCA. In agricultural systems, the availability of quality data for the LCI is a challenge because a large amount of information needs to be gathered. The robustness of the indicators to be calculated will greatly depend on the accuracy of the data provided. Usually LCI data collection will involve interviews and surveys to farmers and advisory organizations, as well as looking for process data from LCI databases (Point, 2008). However, in late years the amount of information collected at agricultural systems in Europe is increasing. Common Agricultural Policy (CAP) aims to support farmers' incomes whilst encouraging them to produce high quality products and to adopt more sustainable practices. In order to follow up CAP objectives, a Land Parcel Identification System (LPIS) has been developed and information systems across the EU hold more than 135 million detailed land parcels, annually declared by 8 million farmers (JRC, 2014). At the Spanish level, the geographic information system for the management of CAP is known as SIGPAC. This system can be used to incorporate information about soil, crop, farming operations, etc., which can then be stored at the parcel level, thus providing unique data for different purposes. Moreover, climatic, meteorological, phenological, etc. data can be assigned to each parcel from the network of meteorological stations and from records in agricultural experimental stations. Using this system, Decision Support Tools (DST) can be developed in order to provide farmers with agricultural advice for the main agricultural operations.

Although the opportunity is arising, there are not many examples linking GIS with LCA, and the ones available provide only a partial integration. Falcucci et al. (2012) incorporated climate and agro-ecological zones in explicit GIS layers for elaborating LCA of greenhouse gas emissions from global livestock production. Hercule et al. (2012) studied how to use GIS data layers to standardize and simplify the choice of inventory for agricultural production: pumping energy required for irrigation, fuel inputs for cultivation (largely determined by clay content of the soil), etc.

As already mentioned, farmers need to collect information for different purposes such as to abide European regulations, to apply for CAP aids, etc. At European level a framework for achieving a sustainable use of pesticides has been developed (EC, 2009). Among the mandatory actions to be implemented, record keeping of any use of pesticides is included. Pesticide use, date, product and amount have to be recorded, so this information can be declared if the farmer is required to do so. Besides, the eco-conditionality aids for small farms are becoming more and more important, since the farmers' income is going to depend increasingly on aids, especially with decoupling. All this translates in record-keeping and report elaboration to benefit from these measures. In this sense, farmers will require standardized systems for collecting information and preparing mandatory reports. Besides, another important aspect of European agricultural policies is the recent inclusion of green payments and climate risk management tools proposed for the CAP for the period beyond 2013, which shows the European Commission's willingness to expand this climate component.

In summary, there is an urgent need to develop tools for increasing the efficiencies of food systems at a lower environmental cost. In this sense, precision agriculture at the interplot and intraplot level is gaining consideration for increasing agricultural productivity. The aim of the sigAGROasesor project (LIFE + 11/ENV/ES/000641) is to develop an online tool capable of displaying customised recommendations for extensive agriculture, in real time, and for specific parcels, on the basis of a series of detailed biotic and abiotic variables. Furthermore, it incorporates a set of environmental indicators to make farmers aware of the environmental impacts of their cropping management practices. It is based on GIS methodology to display the cropping units as well as the variables needed to run the DST, which provide agronomic advice to the farmers as well as alerts of disease appearance risks. An additional goal is to develop a DST for calculating environmental indicators to allow farmers and users of the AGROASESOR platform to measure key aspects of sustainable agriculture. It ultimately seeks to help farmers and farm managers to achieve the most efficient and sustainable crop production systems.

2.Methods

2.1. Platform AGROASESOR

The AGROASESOR platform is an on-line services platform, which comprises three pillars on which this expert system and decision-support tool is based: a) the application of modern GIS technologies for the management of geo-referenced information, making use of soil variability, climate and weather information, crop condition, plant health alerts, and biotic and abiotic risks in the decision-making process, b) Web-based Decision

Support Tools (DST) to systematise decision-making at the farm, and c) geo-referenced traceability, as a tool to register and manage historical records of Crop Management Units (CMU). In this way, the system will use all available information associated with each of the parcels of the farm incorporating it into decision making programs. In this way, farmers, advisors and cooperative managers will have an instrument for extensive field crops, which will provide specific advice, with precise handling tips (varieties, fertilization, irrigation, disease risk). With regard to sustainability, a specific area within the platform will allow the calculation of environmental indicators. The information generated by farmers and agricultural advisors and introduced at the CMU level to provide technical recommendations on the DST is the basis for calculating environmental indicators. Furthermore, through maps, algorithms, data tables, etc. we will incorporate the "extra" needed information (Figure 1). Therefore, the AGROASESOR platform will incorporate environmental criteria to guide economic and social farming practices towards more sustainable production models.

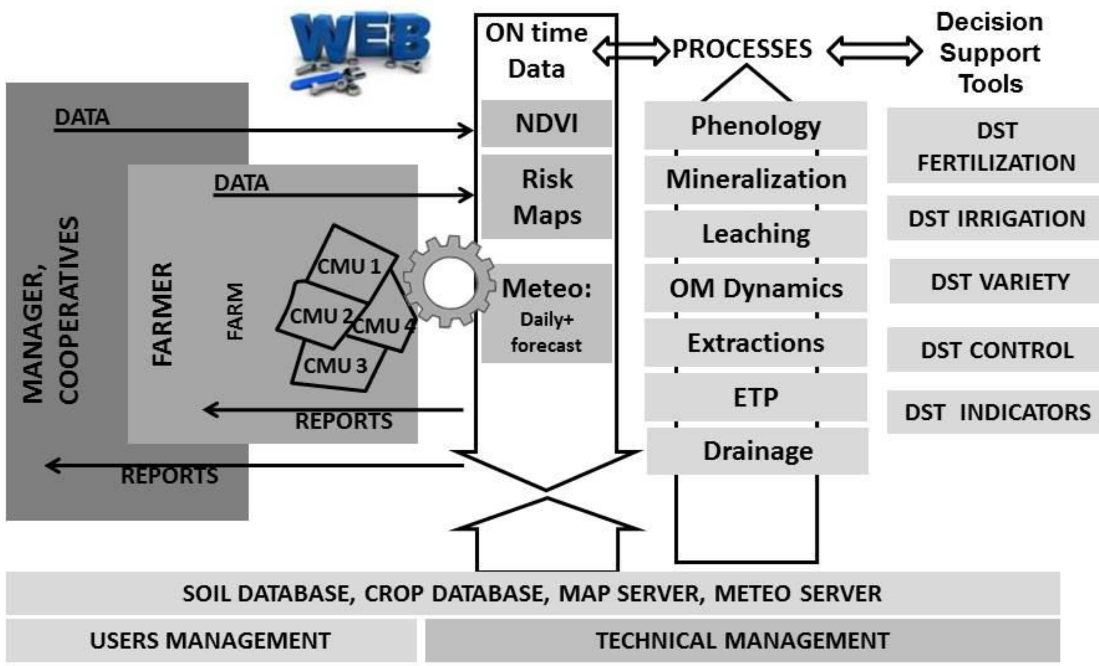


Figure 1. Operating system of the AGROASESOR platform: entities involved in the management of the platform, databases that support Decision Support Tools (DST), general processes performs for obtaining data for the DST. The DST INDICATORS is based on the data of farming operations, including all the inputs involved, as well as outputs in the form of yields from the Crop Management Unit (CMU).

2.2. Development of the environmental indicators at the spatial level

At the AGROASESOR platform, GIS is used to provide information at different levels and scales: a) soil properties (soil depth, texture, SOM, pH, etc.), b) weather variables, climate and meteorological data are assigned to each CMU through an algorithm, which identifies the most representative weather station, c) agricultural inputs and agricultural operations, d) factors needed to calculate the environmental impacts (GHG emission factors related to temperature values, regions with different companies providing electricity, etc.), e) factors needed to assess the environmental impacts, for example, to assess the impact of water consumption in different watersheds.

Besides, we are calculating Gross Margin of the CMU, and we have also selected Soil Organic Matter as an indicator of sustainable land use, which will be followed through time to assess soil quality. In this way we can balance out the different needs that agricultural systems have to take into consideration, maintaining an appropriate balance between economic goals and land maintenance, that is, we simultaneously raise the level of soil organic matter to a certain level and try to maximize benefits from the cropping system.

Carbon footprint calculation is based on a simplified LCA following ISO 14040 standard (ISO 14040:2006) and PAS 2050 (BSI, 2011) methodology; the boundary of the calculation is “cradle to gate”. Upstream processes such as the emissions related to the production of fertilizers and fuel were included, whereas manufacturing and maintenance of machinery and infrastructure were not taken into account within the Scope 3. In relation to the Scope 1, fuel consumption associated to fix and mobile combustion sources, as well as tillage and cultivation practices, were taken into account. Within farming operations the emissions of N_2O from the application of N fertilizers and soil cultivation were calculated using IPCC Tier 1 methodology (IPCC, 2006). Finally, within the Scope 2, electrical consumption was taken into account. Emissions of GHG from the production and distribution of a range of fertilizers, seeds and pesticides were taken from the Ecoinvent database 2.2 (Ecoinvent Centre, 2010). For compost and animal manure, ADEME (2010) and GES´TIM (2010) were used. The average Spanish electricity production mix was selected for calculating the environmental impact of electricity consumption.

Water footprint was calculated following the Water Footprint Network (Hoekstra et al., 2011) and ISO/DIS 14046 (2) methodology, adding its three components: the green, blue and gray water footprint. Although from the LCA standpoint there are some drawbacks in this approach as, for example, not taking into consideration the water footprint associated with the manufacture of raw materials, it has to be mentioned that data from the whole value chain, including all the suppliers, was impossible to get at this stage. Given that,quality changes in different environmental parameters are excluded in the volume based WFN approach and regional aspects are not included in sufficient detail, the Water Scarcity Index (WSI) was calculated as a measure of water use impact. The WSI was calculated based on Pfister et al. (2009), but using the water stress characterization factors values developed for 55 river basins in Spain (Núñez et al., 2013).

Pesticide indicators for pressure (number of applications), ecotoxicity (aquatic and terrestrial) and human toxicity were calculated using the most specific impact values derived for pesticide products. We used 1,4 Dichlorobenzene as an equivalent toxicity unit (Ecoinvent Centre, 2010).

3. Results

The main objective of building the DST INDICATORS at AGROASESOR has been achieved and been tested using a spread datasheet, which was developed previously to incorporate all the required farming operations, data and emission factors. Based on the calculations performed in the spreadsheet, programming was developed to accomplish the needed calculations, while taking into account that a great part of the primary and intermediate data is available in the platform. In this way, users include their management practices, taking into account the tractor, the equipment and the inputs, which are provided in top-down lists; besides, climate, weather, soil characteristics, biotic and abiotic factors are provided. However, the development of such a tool has to find a compromise between simplicity and accuracy, so that results are meaningful for the purpose of the tool and users are not discouraged to use the DST INDICATORS.

3.1. Development of the environmental indicators at the spatial level: assumptions and restrictions

3.1.1. Carbon footprint: Cut-off criteria, assumptions and limitations

The calculator has several sub-systems that decompose overall emissions by greenhouse gas emitted and crop management operations.

3.1.1.1. Fertilization

In the first step the user has to choose the mineral fertilizer, slurry or manure applied from the list that the tool provides. The mineral fertilizers, slurries or manures set out in this list have incorporated specific GHG emission factors from the production and distribution collected from Ecoinvent 2.2 (Ecoinvent Centre, 2010), ADEME (2010), or GES´TIM (2010) databases. Finally, the user has to enter the dose applied at each application. If the applied fertilizer does not appear in the list, the user has to enter the percentage of N, P_2O_5 and K_2O in the fertilizer. In this case, the calculator uses general GHG emissions factors from Ecoinvent 2.2 (Ecoinvent Centre, 2010) database.

Nitrous oxide (N_2O) and nitric oxide (NO) emissions related to fertilizer application were included according to IPCC Tier 1 method (IPCC, 2006), where it is assumed that 1% of applied N as mineral or organic fertilizer is emitted as N_2O. In relation with indirect fertilization-induced soil emissions, leaching is assumed to occur at a rate of 0.3 % of N applied (IPCC, 2006) for a moist climate zone. Although the tool calculates nitrate leaching in the DST fertilization, it is not feasible to use this specific leaching data in the DST INDICATORS in order not to cause an excessive time delay in obtaining the results for the indicators. Emissions of CO_2 from soil resulting from urea application or liming are also accounted for using the IPCC emissions factor (IPCC, 2006) of 0.20 and 0.12, respectively. Finally, the annual amount of N in crops residues (F_{CR}) returned to soil was calculated following IPCC 2006 Tier 1 approach (Eq. 1). The user has to answer the questions of what percentage of the plot is burned and what percentage of the residues is incorporated into the soil. We use the combustion factor (Cf) according to Chapter 2 of IPCC (2006):

$$F_{CR} = \sum_{T} \left\{ \begin{array}{l} Crop_{(T)} \bullet \left(Area_{(T)} - Area\,burnt_{(T)} \bullet C_f \right) \bullet Frac_{Renew(T)} \bullet \\ \left[R_{AG(T)} \bullet N_{AG(T)} \bullet \left(1 - Frac_{Remove(T)} \right) + R_{BG(T)} \bullet N_{BG(T)} \right] \end{array} \right\} \qquad \text{Eq. 1}$$

where,
$Frac_{Renew\ (T)} =$ Fraction of total area under crop T that is renewed annually equals one, because all the crops included are annuals.
$R_{AG(T)}$ = Ratio of below-ground residues dry matter, for cereals = 1.3, based on Spanish National Inventory of Atmospheric emissions.
$N_{AG(T)}$ = N content of above-ground residues, for cereals = 0.006, based on IPCC (2006)
$Frac_{Remov(T)}$ = Fraction of above-ground residues removed annually. If the answer to the question regarding the incorporation of crop residues into the soil is Yes, then we assumed that the factor equals 0, since residues are not removed. If the answer is No, the tool calculates the percentage of removed residues; in this case, we assumed that residues that are not burned are removed.

3.1.1.2. Pesticides

Although at usual application rates of pesticides the GHG emission from their fabrication and distribution is small, we wanted to give special emphasis to this section due to the importance of the effect of pesticide products on toxicology indicators as explained below. In this case, the tool provides a list of more than 2,000 pesticides based on the list published by the Spanish Ministry of Agriculture, Food and Environment, with all the trade names of the authorized pesticides. We compiled emission factors from Ecoinvent 2.2 (Ecoinvent Centre, 2010) for all the different active ingredients. For each trade name, their active ingredients were compiled and, when available, specific emission factors were incorporated into calculations; otherwise, we used the overall emission factor for insecticide, herbicide or fungicide.

3.1.1.3. Direct energy usage

To estimate the fuel consumption of machinery for farming operations such as tilling, drilling, seeding and harvest, we developed a list with the most common machinery used by farmers and we assigned to each machine fuel consumption (L ha^{-1}) according to the studies of the Spanish Institute for Diversification and Energy Saving (IDAE, 2005). Emissions of CO_2 from fuel consumption are accounted for using Ecoinvent 2.2 (Ecoinvent Centre, 2010) emission factors, of 3.066, 2.660 and 3.339 for diesel, petrol and biodiesel, respectively.

Only electrical consumption in irrigation systems has been considered (Sprinkler, 1.0 kWh; Pivot, 0.5 kWh and Drip, 0.3 kWh). We estimated the electrical consumption for each irrigation system according to Guide des valeurs Dia´terre v. 1.11 (2011). Specific emission factors of electricity were taken from the Spanish electricity production mix.

3.1.2. Pesticide application intensity, ecotoxicity and human toxicity

We build on Ecoinvent 2.2 (Ecoinvent, Centre, 2010) to get 1,4-DCB (kg eq.) factors for all the active ingredients available in this database (Table 1). These factors have been applied to the list of 2,000 authorized pesticides in Spain to calculate freshwater, marine and terrestrial ecotoxicity as well as human toxicity for each commercial pesticide.

Table 1. 1,4-DCB (kg eq.) factors for the different active ingredients available in Ecoinvent 2.2.

Active ingredient	Freshwater ecotoxicity	Seawater ecotoxicity	Terrestrial ecotoxicity	Human toxicity
2,4-D	0.058	0.064	0.002	3.747
Alacloro	0.132	0.111	0.005	6.696
Atrazina	0.084	0.074	0.002	4.321
Carbofuran	0.084	0.104	0.004	5.975
Cianacina	0.091	0.079	0.002	4.657
Dicamba	10.672	0.717	0.174	7.745
Diuron	0.106	0.113	0.003	6.376
Glifosato	1.304	0.199	0.026	14.855
Linurón	0.106	0.113	0.003	6.376
Maneb	0.209	0.199	0.001	13.356
MCPA	0.058	0.064	0.002	3.747
Metolacloro	0.397	0.058	0.002	3.112
Paratión	0.505	0.110	0.010	7.275
Propacloro	0.196	0.267	0.023	16.234
Aclonifen	0.101	89.977	0.008	119.040
Captan	0.031	0.036	0.001	2.019
Clorotalonil	0.033	0.042	0.001	2.444
Clorotolurón	0.053	0.072	0.003	4.135
Dimetenamida	0.151	0.152	0.003	8.641
Folpet	0.028	0.034	0.001	1.916
Fosetil-al	0.426	0.076	0.008	4.939
Isoproturón	0.045	0.061	0.002	3.401
Mancozeb	0.209	0.199	0.001	13.296
Mecoprop	0.046	0.051	0.001	2.927
Metaldehído	0.009	0.010	0.000	0.512
Metamitrona	0.064	0.066	0.001	3.865
Napropamida	0.077	1.227	0.146	72.502
Orbencarb	0.086	0.094	0.002	5.660
Pendimetalina	0.017	0.019	0.000	0.945
Prosulfocarb	0.052	0.068	0.002	3.789
Fungicide	0.112	0.091	0.002	5.219
Herbicide	0.132	0.111	0.005	6.696
Insecticide	0.350	0.167	0.006	8.053

Moreover, given the importance of reducing the number of applications, a second indicator is proposed: pesticide pressure defined as number of pesticide applications per plot.

3.1.3. Water Footprint

Taking into account that the purpose of sigAGROasesor is to encourage users to calculate sustainability indicators, in this first step, we have chosen a simplified calculation of water footprint to make it friendlier, as it has been mentioned in the Methods section.

The green water footprint is an indicator of the human use of so-called green water. Green water refers to the precipitation on land that does not run off or recharges the groundwater but is stored in the soil or temporarily stays on top of the soil or vegetation. The tool calculates the green water footprint according to the following equation (Eq.2):

$$WFgreen = \Sigma Pe/production\ (m^3/t) \quad \text{Eq.2}$$

where, Pe = The sum of the effective precipitation from seeding to harvest. These data are calculated on a daily basis from the meteorological data entered into the platform.

The blue water footprint is an indicator of the consumptive use of so-called blue water, in other words, fresh surface or groundwater. The tool calculates the blue water footprint by dividing the amount of the consumption of irrigation water by the production (m^3/t).

The gray water footprint is defined as the volume of freshwater required to assimilate the load of pollutants, based on natural background concentrations and existing ambient water quality standards. This water footprint is calculated by dividing the pollutant load (L, in mass/time) by the difference between the ambient water quality standard for that pollutant (the maximum acceptable concentration Cmax, in mass/volume) and its natural concentration in the receiving water body (Cnat, in mass/volume) and by the production. We assumed: L = Pollution load as N loss by leaching and runoff multiplied by the excess N. On one hand, we estimated a loss rate of 10 %. On the other hand, we calculated the excess of N as the difference between the nitrogen supplied (organic and mineral fertilization, and N supplied by irrigation water) and extractions. These values are compiled specifically from the DST fertilization.

- Cmax = It has an established legal maximum of 50 mg $NO_3\ L^{-1}$, according to the Water Framework Directive (EC, 1991).
- Cnat = 0, because of natural concentrations are not known precisely but are estimated to be low, and thus for simplicity one may assume Cnat = 0.

Finally, the sum of green, blue and gray footprints is the total water footprint.

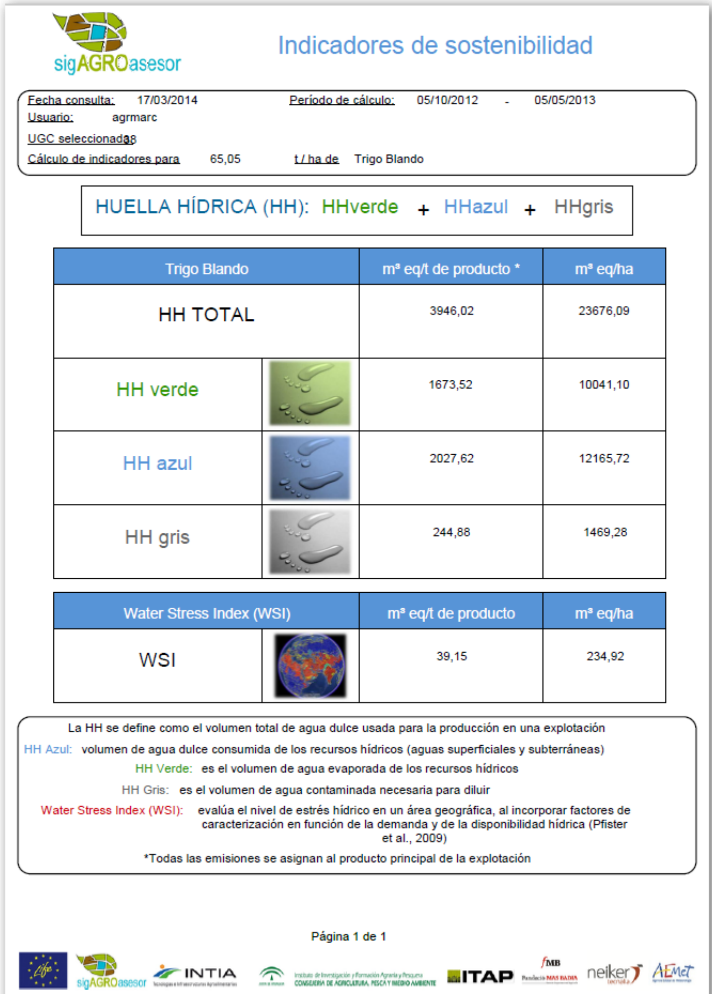
3.1.3.1. Water Stress Index

Water stress is a condition where an imbalance occurs between water demand/need and water availability consumed for meeting the need (UNESCO, 2009). Determination of water stress in an area is calculated using the so called Water Scarcity Index (WSI). We used these values for 55 river basins in Spain as defined by Núñez et al. (2013). The Water Stress Index is calculated by multiplying the water scarcity index for a given area by the water footprint.

3.2. Reporting of the environmental indicators

Once the indicators have been calculated, the tool generates a report with the results for each indicator. As an example, the Water Footprint report is depicted in Figure 2.

In the case of Carbon Footprint, the report indicates total values and the contribution of each of the points to be taken into account (emissions from soil management, emissions from fabrication and distribution of fertilizer and pesticides, emissions from gasoil and electricity). For the Water Footprint, the result is expressed as the sum of the green, blue and gray water footprint and each one separately. In addition, it includes the result of the WSI indicator. Finally, in the case of pesticides, the report expresses the freshwater, marine and terrestrial ecotoxicity as well as the human toxicity and pesticide application intensity.


sigAGROasesor

Indicadores de sostenibilidad

Fecha consulta: 17/03/2014 Período de cálculo: 05/10/2012 - 05/05/2013
Usuario: agrmarc
UGC seleccionada: 38
Cálculo de indicadores para 65,05 t / ha de Trigo Blando

HUELLA HÍDRICA (HH): HHverde + HHazul + HHgris

Trigo Blando	m³ eq/t de producto *	m³ eq/ha
HH TOTAL	3946,02	23676,09
HH verde	1673,52	10041,10
HH azul	2027,62	12165,72
HH gris	244,88	1469,28

Water Stress Index (WSI)	m³ eq/t de producto	m³ eq/ha
WSI	39,15	234,92

La HH se define como el volumen total de agua dulce usada para la producción en una explotación
HH Azul: volumen de agua dulce consumida de los recursos hídricos (aguas superficiales y subterráneas)
HH Verde: es el volumen de agua evaporada de los recursos hídricos
HH Gris: es el volumen de agua contaminada necesaria para diluir
Water Stress Index (WSI): evalúa el nivel de estrés hídrico en un área geográfica, al incorporar factores de caracterización en función de la demanda y de la disponibilidad hídrica (Pfister et al., 2009)
*Todas las emisiones se asignan al producto principal de la explotación

Página 1 de 1

Figure 2. Water Footprint Report (in Spanish).

4. Discussion

One difficulty in the use of LCA, and corresponding environmental indicators, is the interpretation of the environmental impacts because results encompass different environmental impact categories. Methodologies as environmental footprint, which incorporates four environmental indicators: pesticides, greenhouse gas emissions, eutrophication and acidification, have been proposed (Lillywhite, 2008) to overcome the difficulties in interpreting LCA results; at the same time, aggregated indicators are useful for comparing different agricultural commodities in an easier and more comprehensive way. However, the main aim of the AGROASESOR tool with regard to sustainability is to inform farmers on what agricultural operations are causing the largest environmental impacts. So far, as above mentioned, we have incorporated three main indicators to the AGROASESOR platform: carbon footprint, water footprint, and pesticide ecotoxicity and human toxicity. In this way we account for the main environmental impacts of agricultural production. However, interpretation of the results is needed. Our approach is to place the results obtained in a relative scale, which can be built from literature values, or from the values that result from the use of the AGROASESOR platform by the farmers. In this way, a farmer can know how much impact causes in relation to other farmers in a similar production agroecosystem or in a more general scale. This will be helpful for the farmers to decide in which aspects of their production scheme to make favourable changes for the environment. At the same time, the AGROASESOR platform calculates an economic indicator, gross margin, and the farmer can also decide on the economic impact of the assayed agricultural operations.

In a cradle-to-grave analysis, trade-offs between improvements at one stage and increased impacts at another stage are identified (Cowell, 1999). However, AGROASESOR follows a cradle-to-gate approach, considering yield at the field border (CMU) as the product. This fact is partially overcome by assessing several environmental indicators, not only one as is the case for calculations of carbon footprint to measure the global warming potential, which is the objective of many studies.

We calculate indicators in a per production unit as well as in a per surface unit. We can look at a methodology to balance out these two types of calculations. The use of agricultural inputs, in particular, improved varieties, fertilizers, water, pesticides by increasing production in a synergistic way and at optimum adjusted rates by the AGROASESOR platform will allow increasing yields at the same time that decreasing agricultural inputs. In this way the impact per unit of product is expected to diminish. However, when considering impacts by surface unit, we get additional information on absolute amounts of impacts, regardless of production. Thus, we can choose absolute threshold values we do not wish to surpass, and also check that we abide the legislation regarding to the use of agricultural inputs or operations for a given geographical area. This is another contribution to the sustainability of the different production systems. Among the expected impacts of the project are the increase in yields (5%), fertilizer, water and pesticide product use efficiencies (+5%), while decreasing use in absolute values of irrigation water (-5%) and energy consumption (-5%). As mentioned above, Soil Organic Matter has been chosen as a reliable indicator of soil quality and the inner impact of agricultural production.

One of the advantages of the program with regard to environmental indicators is that when asking for reports, the level of aggregation for each indicator can be chosen: one CMU, a set of CMU, all the CMU with a given crop, all the CMU belonging to a farm, etc. In this way it is very easy to establish comparisons to take into account the interest of the farmer or the advisor. At the same time the administrators of the system can work in an integrated approach to answer more general questions regarding crops, inputs, or agricultural operations. Data variables and results can be exported through “csv” files, so by choosing the adequate parameters we can calculate environmental impacts for the variables we are interested, for example, comparing farmers that apply mineral fertilization with those that use mineral fertilizers combined with organic amendments. In this way, we can identify the effect of specific agricultural practices on whatever of the indicators we are interested in.

5. Conclusion

The AGROASESOR platform will allow the extension of the use of sustainability indicators to farmers. This is so because they will have available a web GIS software, which will facilitate the accomplishment of legal, administrative and technical requirements, at the same time that will provide them with the determination of environmental and economic indicators. As all the information needed for the different DST to work is introduced, calculations for the indicators are made easier, thus providing farmers and their advisors with a tool with multiple uses. Further inclusion of models to account for key processes is needed. Furthermore, it would be important to include more crops and indicators to account at the greatest extent with trade-offs at the agricultural level.

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Water in an LCA framework – applying the methodology to milk production in Finland

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ABSTRACT

An LCA case study for water-related impacts on milk production in Finland is presented. Non irrigated feed crops were used in this "cradle-to-dairy" system. Impact assessment used methodology developed by Pfister et al. (2009). Result show that 5.75-5.85 liters of consumptive water is used to produce one liter of packaged skimmed milk. Results from previous studies range from about 1-250 l/kg. Drinking water for dairy cattle is the most significant factor in water consumption (44-45%). Also the volumes of washing water on farms and at dairy factories are substantial. Working practices and equipments affect water consumption on dairy farms, and variation among farms is considerable. Characterization factors for Finland differed significantly from the corresponding factors for geographically and socio-economically similar countries with abundant water resources. The characterization factors concerning Finland should be evaluated and developed to enable more accurate assessments of water-related impacts of products.

Key Words: life cycle assessment, water footprint, milk, animal production, environmental impact assessment

1. Introduction

The global water crisis demands that researchers include water issues in environmental impact assessments of products, particularly food products, which represent most of the world's water consumption. Two evolving methodological frameworks are helping practitioners conserve water resources (Boulay et al. 2013). Originally the water footprint referred to a method developed by Chapagain and Hoekstra (Chapagain et al. 2006, Hoekstra & Chapagain 2007) grounded in the concept of virtual water (Allan 1993, 1997). The evolving ISO Standard ISO/DIS 14046 on water footprint is based on a life cycle assessment (LCA) framework. Both methodologies take account of total direct and indirect water consumption and indicate its relevance (Boulay et al. 2013).

According to Shiklomanov and Rodda (2003), 66% of world water withdrawals and 86% of its consumption are for agriculture. At the local level the share of agriculture is much less in developed countries and in areas with rain-fed agriculture. It also heavily depends on whether imported foodstuffs are taken into account. In Finland agriculture is rain-fed, and the share of agriculture is only 3%, domestic use comprises 25% of water withdrawals and industrial withdrawal is 72% (FAO 2013). Only about 23% of Finland's freshwater consumption (technically introduced freshwater, "blue water") is based on national water resources, mostly because of virtual water content of imported foodstuffs (Mekonnen & Hoekstra 2011).

In the future, as a result of population and economic growth, the global demand for freshwater will increase by 40% more than the available water resources (McKinsey 2009). It was forecasted a decade ago (Shiklomanov & Rodda 2003) that in 2025 about 30-35% of the world's population will live in areas where water availability is very low or catastrophically low, at less than 1000 m^3 freshwater annually per capita. The relative water consumption of agriculture is likely to decrease, mostly due to growth in the industrial and domestic sectors. It is expected that by 2025 global withdrawals for agriculture will increase by 25%, for industry by 50% and for the domestic sector by 80% (Shiklomanov and Rodda 2003).

For these reasons everything possible should be done to ensure that food is produced without consuming local scarce freshwater resources in an unsustainable way. In Finland in practice all field crops are rain-fed, irrigation in the open is mainly used only for vegetables and potatoes (Tike 2012). Finland is one of the EU countries with a Water Exploitation Index WEI of less than 10% which is defined as a "no water-stress"-area (EEA 2003). In terms of water consumption, Finland might be a very good area for water-intensive food production. However, some other water-related environmental problems are very serious. In agriculture in Finland water eutrophication is considered worst environmental problem (Niemi and Ahlstedt 2013) and Baltic Sea is considered to be one of the world's most polluted seas (HELCOM 2007). Because of this, when assessing the environmental impact of food production, it is necessary to consider several water-related problems simultaneously.

In this paper we assess the data availability and quality and also methodological suitability for assessing water-related problems in circumstances of rain-fed agriculture in less used impact categories of water deprivation, and the impacts on human health, ecosystems and resources. We use milk as a case study because milk is, with other animal products, one of the foodstuffs that represent relatively high freshwater consumption during its production chain.

2. Materials and methods

2.1. Milk production system - background

The dairy industry and meat processing are the two main sectors in the Finnish food industry. Dairy products are also the most significant single product group in Finnish food exports, with cheese and butter representing 16% of the total food exports in 2012. Nevertheless, domestic milk consumption exceeds production (Niemi and Ahlstedt 2013). In 2012, still 50% of Finnish dairy cows were kept in tie-stall barns that are gradually being replaced by bigger free-stall barns, as the size of dairy farms increases (ProAgria 2012). More than 80% of Finnish beef production is based on animals originating from dairy herds (MMM 2013).

2.2. System boundary, functional unit and allocation

LCA was used as a framework to assess the water-related environmental impacts of Finnish milk. The system we included:

- feed crop and grass production with agricultural inputs fertilizers, lime, seeds
- industrial feed production
- dairy farms
- dairies, package production and municipal waterworks
- fuel and electricity included in all stages
- transport

Retailers and consumers were excluded from the study.

The functional unit (FU) was one liter of skimmed milk transported from the dairy to the retailer.

Allocation between milk and meat was done according to IDF (2010) methodology for carbon footprint calculation, when the allocation for milk was 87% and for meat 13%. Also allocation between dairy products was done according to IDF methodology, where different weighting factors are provided for electricity, heat, water and raw milk. Raw milk was allocated between dairy products in the basis of the milk solids. (IDF 2010)

2.3. Dairy farm model

Technologies, size of the herd and feed systems vary among dairy farms. We used a dairy farm nutrient management model (Nousiainen et al. 2011) to model fodder consumption in three different sized and technologically different farms (Table 1), where milk production was 8900 kg per dairy cow per year and dairy cow live weight 600 kg (ProAgria 2010). Feeding was based on grass silage, rape seed meal, and either home-grown grain from the farm or a commercial feed. Energy consumption in different farm types was based on figures from the literature (Posio 2010; Edström et al. 2005).

Table 1. Farm types in dairy farm model.

	Farm type 1	Farm type 2	Farm type 3
Technology	Tie-stall dairy barn, pipe milking	Free-stall dairy barn, milking parlor	Free-stall dairy barn, 2 voluntary milking robots
Animals	25 dairy cows, 10 heifers, 9 calves	60 dairy cows, 24 heifers, 22 calves	120 dairy cows, 48 heifers, 42 calves
Feed	Grass silage, own grain and rape seed meal OR Grass silage, commercial feed and rape seed meal	Grass silage, own grain and rapeseed meal OR Grass silage, commercial feed and rape seed meal	Grass silage, own grain, commercial feed and rape seed meal

We modeled all farm types for Southern, Northern, Eastern and Western Finland and combined farms according to statistics to average regional milk production from different sized farms (Table 2). Regional raw milk goes to a dairy located in the same region. Input-output data for different dairies was obtained from the dairy in question. Average Finnish milk production was modeled in relation to size of different dairies. The share of commercial feed (BIF) feeding regime was assumed to be used in 75% of total milk production and on-farm produced feed (OFPF) in 25%, based on expert opinion (Nousiainen 2014, personal communication).

Table 2. Farm types in four regions in Finland (Tike 2012)

	25 dairy cows	60 dairy cows	120 dairy cows
Southern Finland	69%	24%	7%
Western Finland	69%	24%	7%
Eastern Finland	73%	21%	6%
Northern Finland	71%	25%	4%

2.4. Feed production

According to the 2010 Finnish Agricultural Census (Tike 2012) the irrigated area was 0.07% for cereals, 0.03% for turnip rape and rape, and 0.2% for grasses (of the total crop area). The results from the 2010 census support the findings from earlier reports (Pajula and Triipponen 2003; Pajula and Järvenpää 2007) based on expert opinion. The only imported component in the dairy cattle feed concentrate was rapeseed meal, which originates from the Baltic countries (personal communication Raisio 2013), where rape is not generally irrigated (FAO 2013). Applying irrigation in Finland would result in bigger and better quality yields, but irrigation is profitable only in horticulture and for special crops. Irrigation in the open thus represents a marginal share of the water consumption in Finland (Tike 2012). In this study we assumed that no irrigation is applied in the Finnish milk production system.
Fuel consumption in crop cultivation was based on literature (Mikkola and Ahokas 2009). Water consumption in commercial feed production is based on data from industrial feed plant (Raisio 2013, personal communication).

2.5. Drinking water

Several formulas for exist estimating of drinking water intake in dairy cows. The most applicable for Finnish conditions was chosen by comparing the estimate given by a formula and the measured water intake values from the experiment of Kuoppala et al. (2004). The equation presented by Castle and Thomas (1970) had the closest values. It has also very easily measurable parameters (Eq. 1).

Drinking water intake (kg/day) = - 15.3 + 2.53 x milk (kg /day) + 0.45 x diet dry matter content (%) *Eq.1*

Water requirements of growing cattle are often expressed as total water intake, for example in McDonald et al. (1988). In this study, environmental temperature was assumed to be 16 °C, so the total water intake of calves less than 6 weeks of age was assumed to be 7.5 kg/kg DM intake and in older cattle 5.75 kg/kg DM intake. The feed water was calculated according to the dry matter content of a typical diet and drinking water by difference.

2.6. Washing water in dairy farms

The average types of machinery and equipment of the three different farm models were assessed based on the Finnish dairy herd recording system (ProAgria 2012). The database covers approximately 75% of Finnish dairy farms.

Water consumption in conventional pipeline milking systems in tie stall barns was assessed based on expert opinion (Mäki et al 2005). Milking parlors and automatic milking in free stall barns were assessed on the basis of Danish measurements (Rasmussen and Pedersen 2004, Lindgaard Jensen 2009) and on Swedish expert opinion (Gyllenswärd 2011). The sizes of the farm tanks and their wash water consumption and other water consumption in milk storage premises was assessed based on various expert opinions (Nyman 2013, personal communication; Korhonen 2013, personal communication and Gyllenswärd 2011). Other wash water consumption for different farm types was estimated on the basis of building regulations (MMM 2012) and German expert information (KTBL 2008).

2.7. Data sources in processing phase

2.7.1. Dairies

Water consumption data are based on data acquired from three dairies representing the entire dairy sector in Finland. Water consumption data include all freshwater consumption in dairy. Consumption includes milk truck washing. Energy consumption data (electricity and heating) were also based on dairy data.

2.7.2. Package production

Regular skimmed milk is sold in Tetra Rex cartons. We estimated the water consumption of package production on the basis of the Stora Enso Skoghall mill water footprint case study (Stora Enso 2011), for which the water consumption was 6 m^3 / tonne liquid packaging board. The inventory result is 0,164 l water / FU for a Tetra Rex carton.

2.7.3. Waterworks

Municipal waterworks produce water of which on average 18% is uninvoiced (Finnish Water Utilities Association 2012). Uninvoiced water includes water use by the fire department and other municipal use. The amount of unused water due to leakage and pipeline breaks varied according to site. Specific data from waterworks in those sites were used. Water losses in pipelines were calculated only for dairies and water losses in actual dairy sites were 12-15%. Dairy farms were assumed to acquire their water from their own well (Sorvala et al. 2006).

2.8. Other data sources

For electricity, fossil fuel production, fertilizers and lime we used inventory data from EcoInvent v. 2.2. Cooling water and water use in turbines for energy production were excluded. Diesel consumption for wood chips and peat production was assessed according to Mälkki and Virtanen (2003), Jylhä (2013) (wood) and Uppenberg et al. (2001) (peat).

2.9. Impact assessment

Environmental impact of freshwater was assessed according to a method developed by Pfister et al. (2009). Damage in three areas of protection was calculated: human health, quality of ecosystem and resources. Damage assessment was carried out according Eco-indicator-99 assessment framework (EI99) (Goedkoop et al. 2001).

Because production was located mostly in Finland and only minor parts in Baltic countries (rapeseed as part of the bought-in feeds), and no irrigation was used in rapeseed production, the same characterization factors were used for each phase in the production chain.

2.9.1. Water stress index and general impact

Water stress is defined by the ratio of total annual freshwater withdrawals to hydrological availability (WTA). The water stress index (WSI) determines the portion of consumptive water used which dispossess other users of freshwater. In the methodology developed by Pfister et al. (2009, Supporting Information), water stress indices for countries range from 0 to 1. WSI for Finland is relatively high at 0.416. At the watershed level most of Finland gets low WSI values, with some exceptions in the Varsinais-Suomi region, in some islands in the Gulf of Finland and in the Helsinki area. According to Pfister (written communication 2014) high total value in Finland is mostly due to water deficiency problems in the Helsinki area in 1960s and 1970s. Since then the problem has been solved by piping water from an external watershed to the Helsinki area.

Global average WSI (0.602) was used in the water footprint (general impact) calculations. The amount of consumptive water was normalized by dividing by 0.602 and expressed in H_2O equivalents (H_2Oeq).

2.9.2. Impacts on human health

For human health the pathway for water shortages for irrigation resulting in malnutrition is included in the method (Pfister et al. 2009). The Human Development Index (HDI) is an important part of the characterization factor $\Delta HH_{malnutrition}$ for human health. HDI is at high level (> 0.88) for Finland, and as such would cause $\Delta HH_{malnutrition}$ to be zero. There are rivers flowing from Finland to Russia and therefore the cross-boundary watersheds need to be included and CF for $\Delta HH_{malnutrition}$ for a country is calculated by counting up the watershed level results. The total CF for $\Delta HH_{malnutrition}$ for Finland is 4.4E -05 [10^{-6} DALY/m^3] (Pfister et al. 2009). Districts adjacent to Finland in Russia (Leningrad area, Republic of Karelia and Murmansk area) are developing, but still the HDI were in all of them <0.88 in 2013 (UNDP 2013).

2.9.3. Impacts on ecosystem

For places where plant growth is water-limited, withdrawals of so-called blue water may finally reduce the availability of green water and decrease vegetation and plant diversity. CF for ecosystem quality ΔEQ is 1.62E-01 [m^2•yr/m^3] for Finland (Pfister et al. 2009).

2.9.4. Impacts on resources

The renewable annual water resource in Finland is 106 km^3 (Shiklomanov & Rodda 2003) and total freshwater withdrawal 1.485% of actual renewable water resources (ARWS) (FAO 2013). In Sweden the corresponding percentage for ARWS is 1.503% (FAO 2013). The characterization factor for damage to resources in Finland is 7.42E-03 [MJ/m^3] and zero for Sweden (Pfister et al. 2009). According to Pfister (2014, personal communication) this is due to same artifact as the relatively high WSI for Finland. For that reason, CF for water resource impact (ΔR) should be zero and the impact on the resource is insignificant. Because of this, and also because there are only few relatively small areas with high WSI in Finland, and dairy production is located mostly in low WSI areas, we used a CF of zero for ΔR.

3. Results

3.1. Water consumption in production chain

Freshwater consumption for production of one liter of skimmed milk was 5.775 liters. In different feed regimes the results were 5.85 liters for cattle using on-farm grain in feed (OFPF) and 5.75 liters for cattle using commercial fodder as part of their feed (bought-in feed, BIF). The most water consuming phases were animal drinking (44-45%), water use in dairies (23-24%) and washing on the dairy farm (15-16%). Other water consumption in primary production included feed production, energy use and transportation and it comprised 6-7% of total use. Other water consumption in processing was respectively 9-10% and included package production, energy use in dairy, transportation and uninvoiced water in municipal waterworks.

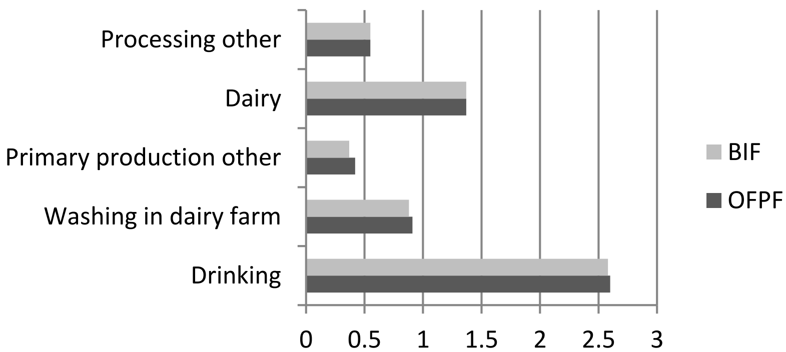

Figure 1. Total consumptive use of freshwater in production of skimmed milk ($liter_{water}$ / $liter_{milk}$) in production systems for on-farm produced feed (OFPF) and bought-in feed (BIF).

Dairy farm water consumption without drinking water was 1.25-1.33 liters/FU. Total water consumption from cradle to farm-gate was varying from 4.36-4.89 liters/FU. Washing the milking machines consumed most of the total amount without drinking water (Figure 2). Water consumption in feed production consisted of water consumption in agricultural input production (fertilizers, lime, fuels, electricity and seeds). After allocation in dairy-phase the total consumption from cradle to farm-gate was 3.8-3.9 liters/FU.

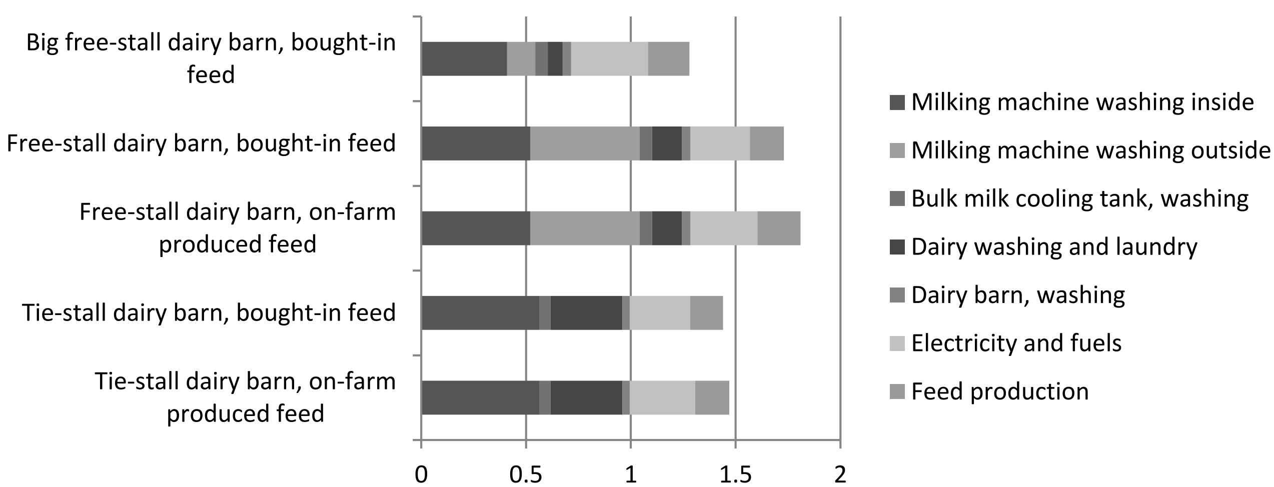

Figure 2. Consumptive use of freshwater in primary production without drinking water ($liter_{water}$ / $liter_{milk\ in\ dairy\ farm\ gate}$).

3.2. Impact assessment results

Impact assessment results were calculated in four impact categories (Table 3). Impact on resources was assumed to be zero (see 2.8.4). If calculated with current characterization factors, the impact would be 4.3E-05 MJ/l,$_{milk}$.

Table 3. Impact assessment results for one liter of skimmed milk.

Consumptive use l/l milk	Water deprivation H_2Oeq/l milk	Ecosystem quality ΔEQ m^2*year / l milk	Human health $\Delta HH_{malnutrition}$ DALY / l milk	Resources ΔR MJ / l milk
5.8	4.0	9.4E-04	2.5E-07	0

4. Discussion

4.1. Discussion of the results

Consumptive water use in milk production based on rain-fed feeds was 5.8 liters / liter skimmed milk. Drinking water for dairy cattle, water consumption in dairies and washing waters on farms were the most significant factors included in total consumptive water. Differences between feeding regimes are negligible in this case. Milking machine type makes a minor difference between farm types, but due to data quality no further conclusions are possible to draw about farm type differences. Numerical results indicate that total consumptive water use in milk production is low and water-related impacts are not in very high level in Finland.

The improvements in methodology (discussion in 4.4) might even decrease the impact assessment results in the future. However, in terms of other water-related impacts, dairy sector in Finland is considered to have high eutrophication impact: in Baltic Sea agriculture constitutes the largest share of the reported total diffuse nutrient loads to the sea (HELCOM 2011).

4.2. Comparisons with other studies

Worldwide water-related impacts of milk production have been studied by several researchers (i.e. Netherlands: De Boer et al. 2013, Australia: Ridoutt et al. 2010, New Zealand: Zonderland-Thomassen, M.A. and Ledgard, S.F. 2012). As shown in Table 4, results vary considerably. This is partly due to different local characterization factors, and as such, indicates the different impacts at various localities.

Production systems are very different, especially regarding irrigation, which affects results significantly. Irrigated feeds were used in the Netherland and New Zealand/ Canterbury cases. In the Netherland the consumptive water use was 66.4 liters in total, but only 5.7 liters without feeds. In New Zealand there was almost a 250-fold difference between the Waikato, with rain-fed feed production, and Canterbury cases.

However, there are several definitions for consumptive water. In New Zealand cases the consumptive water does not include all water withdrawn but water a) withdrawn and then evaporated, b) water returned to another catchment and c) water integrated in a product (see Table 4) (Zonderland-Thomassen, M.A. and Ledgard, S.F. 2012). Other studies include drinking water and irrigation water, if used, in total (De Boer et al. 2013, Australia: Ridoutt et al. 2010), and therefore it is not meaningful to compare the amount of consumptive water.

A huge variation in inventory data also affects the results, especially regarding the amount of water consumption on dairy farms, which varies considerably. In the New Zealand cases, the drinking water was 50 liters / lactating cow / day, in our study 73 liters and in Australia 150 liters. Milk production, feed dry matter and temperature have the biggest impact on cow water intake, and that probably explains the differences.

Table 4. Impact assessment results from previous studies on milk production from cradle to farm-gate. Milk produced by bought-in feed (BIF) in tie-stall dairy barn is used here as an example of Finnish milk.

Study milk from cradle to farm-gate	Consumptive use l/l, milk	Water footprint [3] H_2Oeq /l, milk	Ecosystem quality ΔEQ m^2*year / l milk	Human health $\Delta HH_{malnutrition}$ DALY / l milk	Resources ΔR MJ / l milk
Finnish milk	4.52[1]	3.12	7.32E-04	2.0E-07	0
De Boer et al. 2013 **Dutch milk**	66.4[1]	33.4	1.29E-2	8.0 E-08	6.7E-3
Ridoutt et al. 2010 **Australian milk powder**	14.1[1] (milk) 108[1] (milk powder)	1.9 (milk) 14.4 (milk powder)	-	-	-
Zonderland-Thomassen, M.A. and Ledgard, S.F. 2012 **New-Zealand milk** a) case **Waikato**	0.945[2]	0.165[4]	5.86E-05	0	0
b) case **Canterbury**	249.3[2]	11.1[4]	3.06E-02	0	0

[1]consumptive water includes drinking water and irrigation water, if used, as a whole
[2] consumptive water according to Water Footprint Network (Hoekstra et al. 2009, 2011) includes water withdrawn and then evaporated, water returned to another catchment and water integrated in a product
[3] Water footprint according to Ridoutt et al. (2010)
[4] blue water abstracted

Finnish milk seems to be associated with rather low consumptive water use. However, water footprint is higher than in Australia and New Zealand Waikato, because of the high WSI for Finland. Human health impacts exist, according to the method of calculation, in Russia. Both of these results are significantly affected by CF.

In general the huge variation in results from different studies indicates that the method of calculation is yet not completely established. Because of the local characteristic of water-related problems, it is natural that variation among inventory data is considerable. However, some differences may be due different definitions of consumptive water, different methods of data collection and also in cases of less-qualified data, as in this study (see 4.2), the likelihood of error increases. In addition to different inventory data sets in different studies, there might be differences in system boundaries and allocations. At the moment researchers have still only very few previous studies to compare with their own results.

4.3 Data availability

Good quality inventory data were not always available during inventory analysis. Data variations in other milk studies indicate this situation is not unusual, even though part of the variation is due to natural differences in production systems. Dairies measure their total water consumption, but no product-specific or production-line-specific data were available and therefore allocations were necessary. Water consumption in dairy farms had to be assessed by reviewing the literature thoroughly and canvassing expert opinion. There were some published data on water consumption using specific milking equipment and other equipment, but most studies used as data sources focused on research questions other than freshwater consumption. This definitely had a negative impact on data quality in this context.

Many dairy farms have their own well and no water meter making the assessment of water consumption a challenging task. Because it seems that in many cases that water consumption on dairy farms is very variable, depending on machine control and special practices, e.g. in frequency of washing, a more comprehensive study with rigorous consumption measurements would give interesting information.

4.4. Methodological gaps

4.4.1. System boundaries and allocation

The definition of consumptive water use seems to be very important question in terms of system boundary definition. Irrigation and animal drinking water are the most important phases of the milk production chain, and the results are different when in those phases all water withdrawals are included in inventory or only evaporated water, water returned to another catchment and water integrated in a product. Similar questions arise also in other phases of the production chain.

Municipal waterworks are not very often included in freshwater consumption inventory studies. We noticed that leakage of freshwater in pipelines has an impact, especially when the pipelines are long, and should be included in inventories. Another interesting case is the products of mining industry: lime and K- and P-fertilizers. There is always rainwater flowing into the mine and there is a need to pump it out. Should this water be included into the study or not? In some cases water is clean and available for many purposes like in lime mining, but not in all cases.

In terms of allocation, milk chain is a pretty complicated chain. At the moment a proper method for allocations in water related issues is missing. IDF (2010) recommends raw milk allocation in dairy in the basis of milk solids in climate change impact category, but regarding to water-issues, this should be re-assessed.

4.4.2 Impact assessment

Currently there are characterization factors that are defined using global models, but enhanced modeling tools for regionalized life cycle impact assessment are needed (Pfister et al. 2009). In this study we saw how global models may lead to misinterpretations at the local scale, as discussed in 2.8. In Finland, WSI and CF especially should be assessed again using up-to-date local data for impact their impact on the resource base. Also CF for impact on human health should be redefined, and good data should be found to detail the real impact pathway better to establish if there are connections between health impacts in Russia and water withdrawals in Finland.

5. Conclusion

As a first approach to apply LCA framework to food stuff in Finland, focusing on less used impact categories of water-related impacts, this study provides basic knowledge about suitability of the current methodology and data in this kind of assessments.

Impacts of water consumption in milk production were calculated, but further studies are needed to draw conclusions about water deprivation, human health and water resource impacts. At least WSI and CF for Impact on Resources should be redefined for Finland, using up-to-date local data instead of global models. Human health impact is due to the lower HDI in Russia, which affects the calculation, but there might be reasons to reassess the impact pathway.

Globally, variation in water-related impacts of milk production systems is huge due to location of production and i.e. production type (irrigated versus rain-fed agriculture) and the first thing to do is to point out hotspots of the production chains and magnitude of impacts. LCA framework is well suitable for assessing the consumptive water of food stuffs. We found out that consumptive water of milk production, in Finland, is relatively low. At the moment weaknesses in methodology and CFs based on global models are setting limitations for the impact assessment as well as poor quality of the data. Therefore, both a) improvements in methodology and b) more experience in water-related impacts included in LCA to gradually improve quality and quantity of the data are necessary. LCA is a strong tool to show several impact category results at once and to provide suitable quantitative information for improving the sustainability of the production chains, also when water-related impacts are in question.

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A framework to assess life cycle nitrogen use efficiency along livestock supply chains

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ABSTRACT

Due to the significant contribution of the livestock sector to nitrogen (N) losses, improving N use efficiency (NUE_{-N}) along the life cycle of livestock products is one of the important step towards increasing production performance and reduction of its environmental impacts. We developed a comprehensive framework and novel metrics to assess NUE_{-N} along the livestock supply chain (i.e. "cradle-to-primary-processing"). Our framework was illustrated for the case study of mixed dairy production in Western Europe. Metrics developed included the life cycle NUE_{-N}; total N losses to the environment per unit of N in the final co-products; and the N hotspot index (NHI_{-N}), defined as the relative evenness of the N losses along the supply chain. Averaged across countries, the life cycle NUE_{-N} was 36±3.1%, N losses were 6.6±1.8 g N per g N in the final animal co-products, and NHI_{-N} of 1.0±0.1. The N losses and NHI_{-N} also revealed large differences in hotspots across supply chains, and allowed to identify priority areas where improvement actions are necessary to enhance the efficiency. We show that the combination of life cycle NUE_{-N}, N losses and NHI_{-N} gives valuable information to guide N management in livestock supply chains.

Keywords: life cycle, nitrogen use efficiency, hotspot, livestock, supply chain

1. Introduction

The livestock sector is identified as a significant contributor of nitrogen (N) losses into the air, water and soil, which can create environmental burdens including climate change, eutrophication, degradation of water and air quality (Erisman et al. 2007; Galloway et al. 2010; Xue and Landis 2010; Leip et al. 2013; Sutton et al. 2013). Due to an increasing world population and changing dietary patterns, driven by rising incomes and urbanization, especially in developing countries, the global demand for livestock products is expected to increase in the coming decades (Alexandratos and Bruinsma 2012). Acknowledging the need to improve its environmental sustainability, the livestock sector is increasing its efforts to improve nutrient management and nutrient use efficiency in livestock supply chains[1].

Several researchers proposed N use efficiency (NUE_{-N}) as a valuable metric to manage and benchmark N use and improve performance (Powell et al. 2010; Leip et al. 2011; Gourley et al. 2012a; Sutton et al. 2013). NUE_{-N} is generally computed at animal level (Powell et al. 2010) or farm level (Aarts et al. 2001; Gourley et al. 2012b; Godinot et al. 2014). Livestock supply chains, however, are increasingly long and complex, implying that NUE_{-N} at animal or farm level captures only a fraction of actual emissions. Potential inefficiencies taking place upstream or downstream from the farm are thus ignored. It is necessary to bring the life cycle approach to NUE_{-N}, in order to identify "hotspots" along the entire supply chain, and to avoid interventions that would result in shifting nutrient inefficiency problems from one production step to another.

Few studies have addressed nutrient use efficiency (NUE) at chain level (Suh and Yee 2011; Wu et al. 2014) and several challenges remain concerning the computation of a representative life cycle NUE_{-N}. The aim of this paper, therefore, is to present a comprehensive framework and related metrics to assess NUE_{-N} along the livestock supply chain, from "cradle-to-primary-processing." The framework will be illustrated with a case study.

[1] This research takes place in the context of a multi-stakeholder partnership gathering private, governmental and non-governmental organizations and aiming at developing methods and metrics relevant to guide the sustainable development of the livestock sector. See Livestock Environmental Assessment and Performance (LEAP) website: http://www.fao.org/partnerships/leap

2. Materials and methods

2.1. Description of the system

We propose a comprehensive method to assess the life cycle NUE_{-N} of livestock supply chains, from cradle-to-primary-processing-gate. This framework can be also used for other nutrients, such as phosphorus. To illustrate the computation and results from this framework, we apply it to mixed dairy systems in 28 countries of Western Europe. Mixed dairy systems were selected because of data availability, and the fact that they offer a relevant mix of inputs and outputs to test the framework. Mixed dairy systems are defined as systems combining dairy farming with other associated agricultural activities, such as feed cropping, other animal species production, and are characterized by an intensive exchange of products and services between these different activities (Oomen et al. 1998).

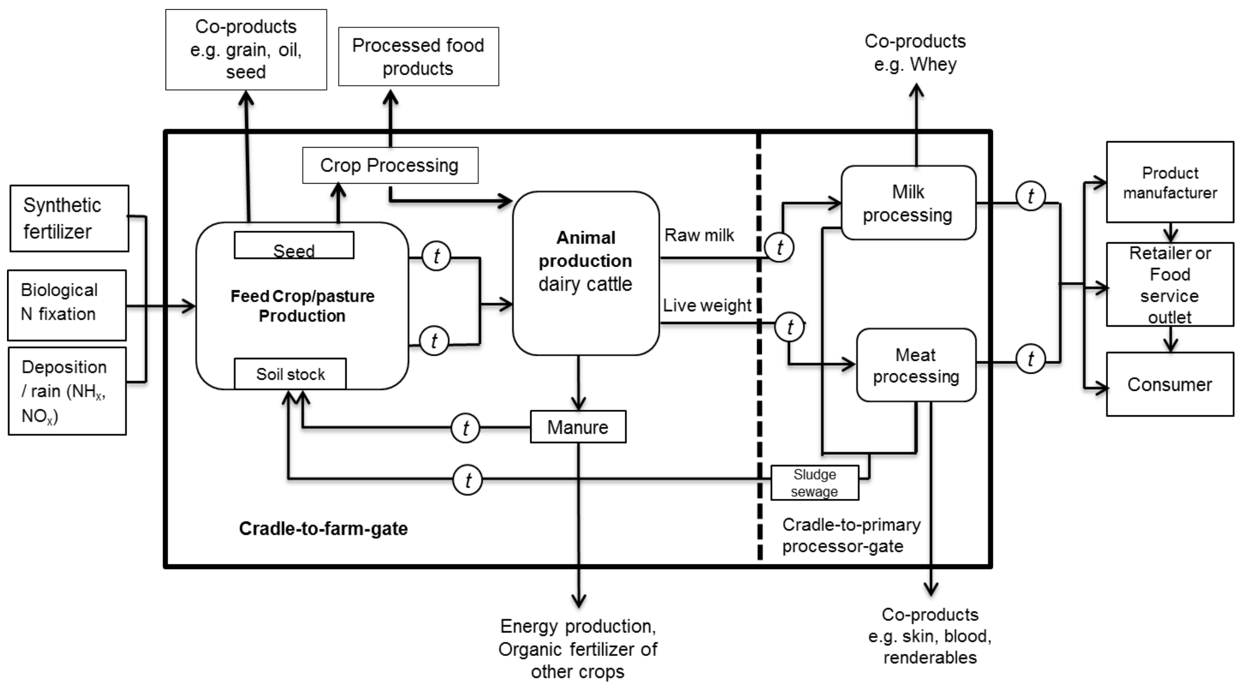


Figure 1. System boundary proposed for the life cycle of dairy cattle covering cradle-to-primary-processor-gate for two main products (milk, meat), the large box covering the cradle-to-primary-processing gate represents the stages covered by life cycle NUE_{-N} methodological framework. (*t*): refers to transportation.

2.2. System boundary

The framework assumes that all N flows occur during the same year, and the system operates in a fixed state. The system boundary illustrated in Figure 1 represents the cradle-to-primary-processing-stage of the life cycle. This choice of the system boundary is motivated by data availability. It includes three main stages, which are interconnected: feed crop/pasture production, animal production, and animal products processing. All N flows entering, leaving or within the system boundary are included in the assessment. Nitrogen flows enter the system mainly through synthetic fertilizer, biological N fixation (BNF), and atmospheric deposition (ATD) to produce feed crop and forage. Other inputs of N to feed crop/pasture are internal to the system and come from recycled manure (urine, dung and litter), crop residues or seeds. N inflows related to fossil fuel are excluded in this study because of lack of sufficient data on national and international transportation of feed stuffs and on-farm fossil fuel use. The N outflows are estimated based on products' yields. We assume that all outflows are separated after their excretion, which imply that, when a food crop, crop residue or by-product is used as feed, an economic allocation factor is only applied to split N losses between crop co-products. At the animal production stage, N flows are associated with feed, milk, live animals and manure recycled. Only milk and live animals are processed into final animal co-products, while manure is used as organic fertilizer in crop production. These final co-products mainly include milk, meat, leather, fat and other non-edible products. At each stage of the supply chain, the fundamental principle of mass balance conservation is applied: "N inflow = N product + N loss + δN stock." N product includes all co-products delivered at each stage, whereas N loss refers to N lost via volatilization, run-off and leaching.

2.3. Metrics

We propose to calculate three metrics based on the analysis of the N flows along the chain, i.e. NUE_{-N}, the N losses and the N hotspot index (NHI_{-N}). NUE_{-N} is a relatively common metric, but it is calculated here for the entire supply chain, following a life cycle perspective. The "N losses" indicate the amount of N lost to produce 1g of N in the final co-product. NHI_{-N}, on the other hand, is proposed as a new metric that qualifies the evenness of inefficiencies along the supply chain.

2.3.1. Computation of the life cycle NUE_{-N}

NUE_{-N} refers to the total N outflow excluding losses to the total N inflow within each stage or subsystem of the supply chain. The calculation model includes three interconnected stages: feed crop/pasture production, animal production and processing, to ensure that an intervention arising from a particular stage of the chain is transferred throughout the supply chain. This concept was previously defined in Suh and Yee (2011) and Wu et al. (2014).

Feed crop and pasture production

At the level of the cropping system, NUE_{-N} is estimated for all crops and pastures produced. NUE_{-Nc} is defined as the ratio of N outputs (harvested in plant biomass) plus the variation of the mineral N in soil stock pool to N inputs (inorganic fertilizers, livestock manure, crop residues, ATD, and BNF). Furthermore, it is assumed that extra N which is not up-taken or lost; is stocked as mineral N in the soil pool. NUE_{-Nc} is estimated as follows:

$$NUE_{-Nc} = \frac{\sum(O_c + \delta S_c)}{\sum I_c} x100 \qquad \text{Eq. 1}$$

where O_c is the total N in harvested crop or pasture; δS_c is the change of the stock of N in soil (either as accumulation or removal of mineral N in soil); I_c is the total N input materials to feed crop/pasture production.

Feed ration

Feed assessment requires the identification of all feed component used in animal ration. We estimate the average NUE_{-N} of ration (NUE_{-Nf}) based on proportion of each feed component in the ration. Feed ration may vary during the year or over a period of a year. It is assumed that animals of the same cohort are fed a similar ration the whole year. In this study, the country specific feed rations are extracted from the Global Livestock Environmental Assessment Model (GLEAM) (Gerber et al. 2013). NUE_{-Nf} is estimated as follows:

$$NUE_{Nf} = \sum_{i=1}^{k} (NUE_{Nci} x \, \beta_i) \; (k = 1,2, \ldots, k) \qquad \text{Eq. 2}$$

where NUE_{-Nci} is the N use efficiency of feed component i, β_i is the proportion of feed component i in the ration, k is number of feed component used. NUE_{-Nf} is calculated at herd level.

Animal production

At the level of the animal production system, NUE_{-N} refers to the capacity of animals to incorporate N in feed into animal products. After the ingestion, N intake is partitioned into edible protein (e.g. meat, milk) and inedible protein (e.g. leather, blood, offal) as well as in animal excreta according to Powell et al. (2013). We considered manure as a valuable product of the animal production system, contrary to most studies that examine N balances of livestock systems (Segato et al. 2010; Leip et al. 2011; Oenema et al. 2012). N losses associated with manure management are estimated using GLEAM (Gerber et al. 2013). We assume that manure managed in mixed dairy systems is used either as organic fertilizer or for energy production. Similarly to crop production, NUE_{-N} in animal production (NUE_{-Na}) is defined as the ratio of N in co-products to N in inputs (feed intake), and is estimated as follows:

$$NUE_{-Na} = \left[\frac{O_a+\delta S_a}{I_a}\right] x100\% \qquad \text{Eq. 3}$$

where O_a is the N in product including edible and non-edible products such as manure; δS_a is the change of N stock in animal herd related to calving or input of new animals; I_a is the total N intake.

Processing

After the farm gate, live animals and primary animal products are processed. The inedible products from processing may be used as compound feed (e.g. blood, bones) for other animal species, as input to industrial processes (e.g. leather, cosmetics, food or pharmaceutical industries) or other use such as fertilizer. N concentration in manure produced during transportation or during stalling of animals at the slaughterhouses and in waste water, which is not treated at factory level, is considered a pollutant to the environment, therefore, is included in the estimation of N losses. NUE_{-Np} is defined as the ratio of N content in final animal co-products to N entering the processing unit and is estimated as follows:

$$NUE_{-Np} = \left[\frac{O_p}{I_p}\right] x100\% \qquad \text{Eq. 4}$$

where O_p is the total N in final animal co-products at primary processing and I_p is the total amount of N in animal production output which equals to O_a (Eq. 3). It is assumed that no stock exists in processing plants.

Supply chain

The entire life cycle NUE_{-N} is estimated as the product of NUE_{-N} of each stage of the supply chain, and is estimated as follows:

$$\text{Life cycle } NUE_N = \prod_{i=1}^{n} NUE_{-N_i} (i = f, a, p) \qquad \text{Eq. 5}$$

Where NUE_{-Ni} represents NUE_{-N} of stage i of the supply chain.

2.3.2. Nitrogen losses

"N losses" is defined as the total N losses to the environment that occur for the production of one unit of the final co-product. N losses are estimated as N in output products plus stock change minus N inputs. The reference unit used to report the N losses is 1 g of N in final animal co-products at the primary processing gate. N losses may occur through volatilization of NH_3, emissions of N_2O, NO, and nitrates leaching and run-off at every stage of the supply chain. They are estimated based on tier-1 equations from IPCC guidelines (IPCC 2006).

2.3.3. Nitrogen hotspot index (NHI_{-N})

NHI_{-N} is defined as the relative evenness of the N losses in the supply chain. NHI_{-N} is calculated as the standard deviation of N losses divided by the average N losses of all stages of the supply chain, and, therefore, quantifies the relative evenness or concentration of losses along the chain. A high NHI_{-N} implies that the occurrence of major hotspots of N losses in the chain, whereas a low NHI_{-N} implies that N losses are evenly distributed. NHI_{-N} is estimated as follows:

$$\mathrm{NHI_N} = \frac{\sigma}{\mu} \qquad \text{Eq.6}$$

where σ is the standard deviation of the N losses at the different stages of the supply chain; μ is the corresponding average of N losses.

2.4. Data and parameters

Country average data regarding crop and livestock activities were extracted from the GLEAM database. GLEAM was developed to assesses the environmental performance of livestock supply chains at global level (Gerber et al. 2013). GLEAM is a Geographical Information system (GIS) based model which represents processes and activities, from the production of inputs into the production process to the farm-gate. It is composed of five main modules: herd module, manure module, feed module, system module and allocation module. GLEAM relies on IPCC Tier 1 and Tier 2 approaches and emission factors to estimate the greenhouses gas emissions and thus the N flows. The Tier 1 approaches are used in this study for the estimation of N_2O emissions. Additional data on N ATD were obtained from global maps of atmospheric N deposition (Dentener 2006), while BNF data are estimated from Herridge et al. (2008). The fourteen feed components considered were different types of fresh grass, legumes, cereals, and by-products (soybean cake, maize gluten meal, etc.) according to Opio et al. (2013). The data represented the year 2005.

2.5. Sensitivity and statistical analysis

The descriptive statistics and sensitivity analysis were performed with R software (R Core Team 2013). The sensitivity analysis of key data was carried out to identify the most sensitive parameters to the life cycle NUE_{-N} result. Each key data of the input has been changed by 10% and the corresponding change in life cycle NUE_{-N} was evaluated.

3. Results

3.1. Overall NUE_{-N} and N losses at different stages of the supply chain

The country-level NUE_{-N} and N losses of mixed dairy supply chains for Western European countries are summarized in Figure 2. Of the 11 Tg N applied annually to feed crop and pasture, only 0.79 Tg N were found in final animal co-products and 2.5 Tg N were recovered as manure available for recycling. The NUE_{-N} is higher at processing stage: 79% compared to 61% and 76% at the biophysical stages of feed crop/pasture production and animal production, respectively. The lower NUE_{-N} in feed crop/pasture production is related to the relatively large amounts of N lost via volatilization, leaching and run-off during the process of application of synthetic fertilizer and manure and manure deposition during grazing. On the other hand, the relatively higher NUE_{-N} in animal production is explained by the large proportion of manure that is estimated to be recycled. About 3.9 Tg N were lost during feed crop/pasture production representing 74% of total N losses in supply chain. The amounts of N losses in animal production and processing stage were lower: 1.1 Tg N and 0.18 Tg N, respectively.

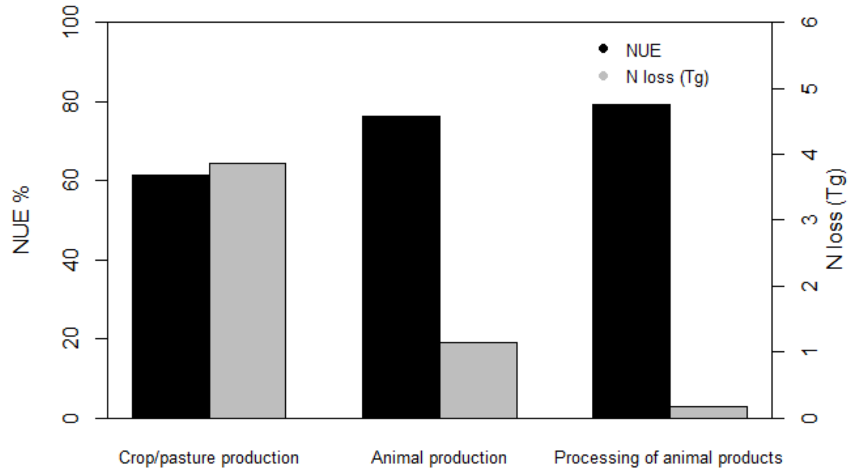
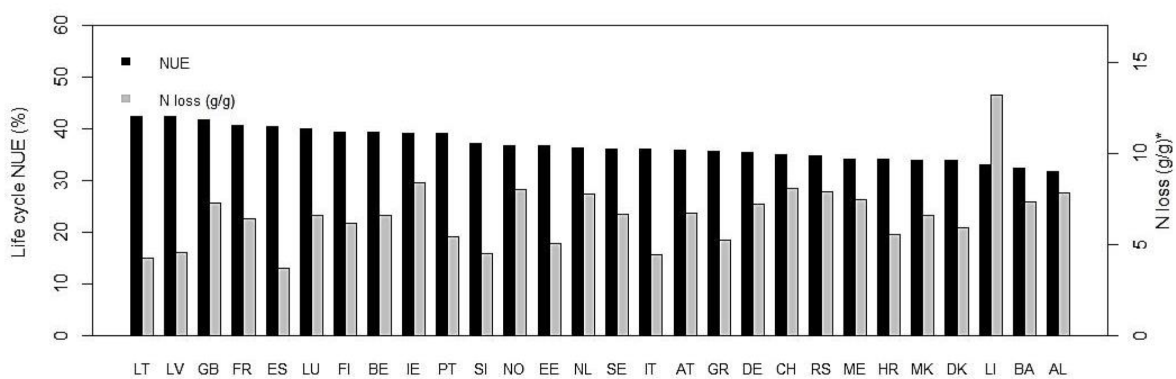


Figure 2. The NUE$_{-N}$ and N losses for every stage of the mixed dairy cattle supply chains for Western European countries.

3.2. Life cycle NUE$_{-N}$, NHI$_{-N}$, and N losses

The average life cycle NUE$_{-N}$ and N losses of mixed dairy supply chains for Western European countries are summarized in Figure 3, whereas results of NHI$_{-N}$ are presented in figure 4. There were large differences between countries in average life cycle NUE$_{-N}$ which ranged from 32% to 43%, with an overall average of 37±3%. Depending on the country, N losses per g of N in final co-product varied from 3.7 g N per g N to 13 g N per g N with an average of 6.6±1.8 g N per g N. NHI$_{-N}$ varied from 0.61 to 1.4, with an average of 1.0±0.1. These results show that the countries with higher life cycle NUE$_{-N}$ do not necessarily have lower N losses per unit of product or NHI$_{-N}$ or vice-versa. For instance, the United Kingdom (GB), and the Spain (ES) have a similar life cycle NUE$_{-N}$ but the dairy cattle system in GB has a higher N losses than in ES. This is because in GB the feed system is dominated by forage (around 62% of total feed), whereas this material plays a less important role in the feed rations of ES (around 42% of total feed) and that fodder production is characterized by a lower feed use efficiency.



AL: Albania, AT: Austria, BE: Belgium, BA: Bosnia and Herzegovina, CH: Switzerland, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, HR: Croatia, IE: Ireland, LI: Liechtenstein, IT: Italy, LT: Lithuania, LU: Luxembourg, LV: Latvia, MK: Macedonia, NO: Norway, NL: Netherlands, PT: Portugal, ME: Montenegro, MK: The former Yugoslav Republic of Macedonia, RS: Republic of Serbia, SE: Sweden, SI: Slovenia.
* N loss is expressed as g N losses per 1 g of N in final animal co-products (edible and non-edible).

Figure 3. The life cycle NUE$_{-N}$ and N losses of mixed dairy cattle supply chains for Western European countries.

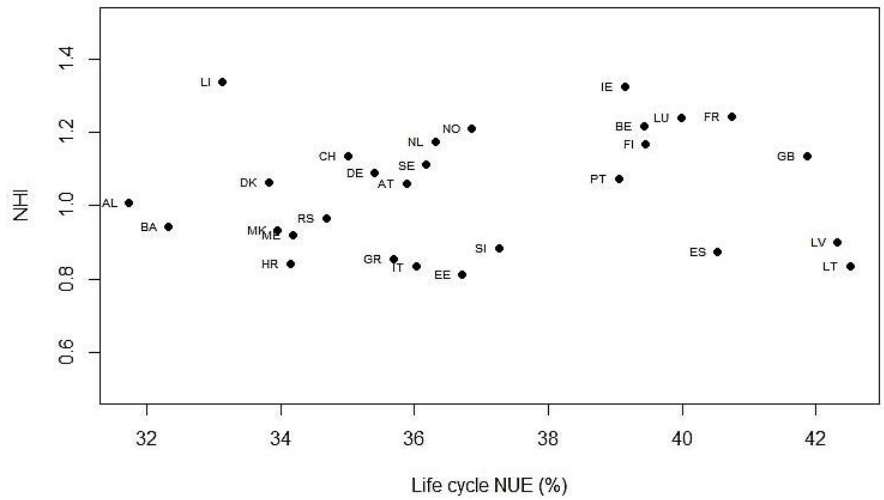


Figure 4. NHI_{-N} and life cycle NUE_{-N} of mixed dairy cattle supply chains for Western European countries.

Around 15 countries have a NHI_{-N} greater than the average (1±0.1), indicating the presence of hotspots in the supply chains. All countries irrespective of their life cycle NUE_{-N}, the feed crop/pasture is the stage with higher N losses due to the over-fertilization. NHI_{-N} also allows comparison of supply chains with equal life cycle NUE_{-N}. In this case, for example, Norway (NO) and Estonia (EE) have similar life cycle NUE_{-N} but the lower NHI_{-N} in EE implies that N losses are similarly spread along the supply chains than in NO where there is a presence of a hotspot in the feed crop/pasture production.

4. Discussion

4.1. Sensitivity analysis and methodological limitations

Data used in this study were mainly extracted from GLEAM and literature. Obviously, these data are subject to various degrees of uncertainty. A partial sensitivity analysis was thus carried out to identify the most sensitive key data to the life cycle NUE_{-N} result. A 10% increase/decrease in the synthetic fertilizer use resulted in a 0.3% increase/decrease in the life cycle NUE_{-N}, whereas it resulted in 1% increase/decrease when applied to feed N intake. On the other hand, the 10% change on manure use resulted in 7% decrease/increase of life cycle NUE_{-N}, which shows the largest range among the key data tested. Any change of this parameter impacts also the NHI_{-N} and N losses. Other sources of uncertainty are related to the modeling of soil mineral N stock change which is simplified in this assessment, and based on a mass balance approach. In addition, the use of Tier 1 and Tier 2 approaches presents a limitation to the accuracy of quantification; more research is needed to develop Tier 3 approaches that improve the accuracy and precision of computation.

The main challenge of computing NUE_{-N} at each life stage of the supply chain compared to computing a unique overall efficiency, is obviously the high data requirements and the dependence on data quality and uncertainty. Therefore, the exploration of minimum data requirement to perform a representative regional or global life cycle NUE_{-N} is crucial. The other limitation is related to the difficulty to estimate mineral N stock in the soil pool. Leip et al. (2011) assumed a soil stock change of zero to simplify the calculation. In contrast, the framework we propose, considers that the extra N which is not uptaken or lost through leaching, run-off and volatilization is stocked as mineral N in the soil pool. The long-term accumulation of N may however result in high leaching with impacts on water quality and aquatic ecosystems. Further improvement of the framework should thus aim at estimating the maximum storage capacity of the soil to avoid the under-estimation of N losses to the environment.

4.2. Comparison with other studies

This study computes NUE_{-N} in a life cycle dimensions of mixed dairy production systems in Western Europe. It shows that, for production of final animal co-products, the life cycle NUE_{-N} was on average 36±3.1%. The country-level life cycle NUE_{-N} computed in this study is not directly comparable with previous results computed at animal or farm level. Nevertheless, partial validation is possible by comparing the NUE_{-N} in different stages of the supply chain with actual quantifications. The average NUE_{-N} at feed crop/pasture production (61%) is comparable to the average of soil NUE_{-N} of 59% estimated for all crops in fifteen countries of European Union (OECD 2001). In animal production, the NUE_{-N} is 76%, which is higher than previous estimates of about 22% in dairy production system (Powell et al. 2013). The difference is related to the N outflows included in the quantification. Powell et al. (2013) only considered the raw milk as output whereas in this study, raw milk, live animals, and recycled manure were included as products, which increased considerably the NUE_{-N} in this stage. The high performance at the processing stage in terms of NUE_{-N} is related to the technological control of flows, treatment of waste water and re-use of organic wastes usually applied at industrial level.

The average N losses calculated in this study are not comparable to the N footprint defined by Leip et al. (2013), which focuses on the total direct N losses to the environment per unit of food product. The N footprint, therefore, is higher than the estimated N losses which are related to the unit of N in the final animal co-product. Furthermore, livestock supply chains deliver non-edible final co-products which constitute raw material for other industries e.g. leather, cosmetics, pharmaceutical industries, etc. Therefore, the harmonization of the quantification methods and interpretation of the results are essential to allow the comparability.

4.3. Relevance and comparative advantages of the proposed framework

The proposed framework relies on the computation of three indicators. Complementarily to life cycle NUE_{-N}, NHI_{-N} facilitates the design of interventions, by giving information on the presence of hotspots. Furthermore, N losses gives information on the likely environmental impacts, although pathways aren’t described in this assessment. At crop/pasture level, this framework may support the improvement of land management, and precision of N application. At the animal production level, increasing feed digestibility and manure management are suggested options. Although further research is needed to explore their applications in contrasting farming systems (Oenema and Tamminga 2005; Erisman et al. 2007; Oenema et al. 2012; Sutton et al. 2013). Relatively high values of NHI_{-N} indicate that interventions can be targeted to few hotspots, which may well improve their cost-effectiveness; whereas designing and implementing interventions for supply chains with lower NHI_{-N} would require greater monitoring and evaluation efforts.

The life cycle approach applied to N use allows to account for the use, recycling and disposal of different agricultural resources such as crop residues and manure. The proposed framework provides indications on potential N pressure on the environment but does not give direct information on the impacts e.g. on water quality. Assessing impacts and evaluating the possibility to use life cycle NUE_{-N} as a potential proxy of impact, requires an approach that takes into account the N migration and transformation pathways in the environment. This requires detailed data on soil types, climate, water catchment, hydrology, drainage capacity of the soil, altitude, intensity of mechanization, as well as the interaction of livestock activities with other agricultural supply chains (Schulte et al. 2006).

Compared to life cycle assessment (LCA), the proposed framework requires less data and assessment steps, while supplying relevant information to support decision making. It is therefore proposed as a tool that can support the design and monitoring of N management in livestock systems that is less data demanding than LCA.

5. Conclusion

The life cycle NUE_{-N} provides information on N performance in livestock supply chains. It explores the capacity of livestock supply chains to convey N contained in inputs into products for human use. The life cycle NUE_{-N} considers the ability of livestock stakeholders to recycle, reuse and dispose of nutrients to limit the continuous importation of “new nutrient” in the supply chain. The combination of life cycle NUE_{-N}, N losses and NHI_{-N} provides relevant information to improve production technology and practices for a better management of nutrients. The framework requires less data than LCA but global and regional assessment would still require much information. Further work is thus needed to evaluate the minimum data requirement of the framework, ensuring wider applicability.

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Environmental footprint of milk production from Mediterranean sheep systems

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ABSTRACT

In the Euro-Mediterranean countries the sheep milk production is a significant sector that is currently going through a deep structural crisis. Since eco-sustainability of production systems and mitigation of climate change are European priorities, optimizing the environmental footprint of dairy sheep farms could become the access key of farmers for receiving financial support and achieve a more environmentally-sound agriculture. The main objectives of this paper were to compare the environmental impacts of sheep milk production from three different dairy farms in Sardinia (Italy), characterized by different input levels, and to identify the hotspots for improving the environmental performances of each farm, by using an LCA analysis. The LCA conducted using different functional units (1 kg of Fat Protein Corrected Milk and 1 ha of Utilized Agricultural Area) and different assessment methods (IPCC, ReCiPe, and Blue Virtual Water) led to a more objective evaluation of the environmental performances of farms.

Keywords: dairy sheep farming systems; environmental impacts; Life Cycle Assessment; agriculture multifunctionality; eco-sustainable livestock.

1. Introduction

The dairy sheep production is a significant sector for the European Mediterranean countries. Sardinia (Italy) is the most important EU region for sheep milk production, with more than 3.2 million ewes – about 3.5% of the EU total (EUROSTAT, 2012) – and a milk production of about 330.000 t year^{-1} (Osservatorio Regionale per l'Agricoltura, 2012), which represents more than 12% of the total European production (EUROSTAT, 2012). More than half of Sardinian sheep milk production is addressed to cheese industry for "Pecorino Romano PDO" (Protected Designation of Origin, European quality label) production (Furesi et al., 2013). "Pecorino Romano PDO" is one of the main Italian PDO product (ISMEA, 2012) and 95% of its production derives from Sardinian cheese factories (Idda et al., 2010). The dairy sheep farming system in Sardinia presents different degrees of intensification, depending basically on the geographical location of farms, which affect key-traits such as arable land availability, soil fertility and possibility for irrigation (Caballero et al., 2009; Porqueddu, 2008). In the last decades, Sardinian sheep production systems suffered a serious and continuous loss of competitiveness, due to several internal and external factors that caused a deep structural crisis in this traditional sector.

As production systems' eco-sustainability and climate change mitigation are on top of the European agenda, minimizing the ecological footprint of farms represents a key factor for farmers to obtaining public incentives and for enhancing the multifunctionality of agricultural systems. Therefore, the optimization of environmental performances could be a crucial factor to improve competitiveness of sheep farming, particularly in marginal areas. For this purpose it is essential to assess the environmental footprints of these livestock systems and to identify the weak points of the production chain where to take actions for reducing the farm's environmental impact (FAO, 2010). The environmental impacts (including greenhouse gases emissions) of animal production systems can be evaluated by using the Life Cycle Assessment (LCA) approach (De Boer, 2003). However, when applied to agriculture, the LCA analysis presents some challenges due to the intensive nature of required data, their limited availability and the multiple-output nature of production (FAO, 2010). Most of the research literature has been focused on intensive livestock systems of dairy cattle (Milani et al., 2011). To our knowledge, very little research studies have been conducted on life cycle assessment of sheep milk. A good example is given by Michael (2011), who made a detailed analysis of the resource use and emissions of sheep milk production in Australia, as part of a wider study on selected "new animal products" industries. On the other hand, several studies have been conducted to evaluate specific impacts of dairy sheep farming systems on soil, water, biodiversity, etc. (Molle et al., 2008; National Research Council, 2010; Peyraud and Delaby, 2006; Shepherd et al., 2007).

This study was conducted with the main aim of contributing to fill in these knowledge and data gaps and with the specific objectives of (i) comparing the environmental impacts of sheep milk production from three different dairy farms located in Sardinia and characterized by different input levels, and (ii) identifying the hotspots to improve the environmental performances of each farm, by using an LCA analysis.

2. Methods

2.1. Case studies

During 2011, data were collected from three different dairy farms located in the Province of Sassari, North-western Sardinia, Italy. The three studied farms fall into a homogenous agro-climatic area, with climate conditions typical of the central Mediterranean area, an average annual rainfall of approximately 550 mm, mean monthly temperatures varying from 10 to 26 °C, and elevation ranging from 60 to 350 m a.s.l. Rural landscape is characterized by dairy sheep farms with a mosaic of feed resources mainly represented by annual forage crops, cereal crops, improved and natural pastures.

The three farms differed mainly in stocking rate, size of grazing areas and concentrates consumption (Table 1), mostly covering the range of input levels for Sardinian sheep livestock (ARAS, 2013). We considered as low input farm (LI), the farm with the lowest stocking rate (1 ewe ha^{-1}), the largest grazing area (95 ha) and the lowest consumption of concentrates (1 t per year). On the opposite, the high input farm (HI) showed the highest stocking rate (5.5 ewes ha^{-1}), the smallest grazing area (12 ha) and an annual consumption of concentrates of about 200 t. Mid-input farm (MI) was characterized by intermediate levels of input. Farms had also different market strategy: LI and HI farms sold the milk to the cheese industry for “Pecorino Romano PDO” production, while MI uses its own milk for small-scale on farm cheese production, “Pecorino di Osilo”, which is included in the Italian list of typical agri-food products. Moreover, MI was the only farm that used the aseasonal lambing technique.

Table 1. Main characteristics of production system in low- (LI), mid- (MI), and high-input (HI) dairy farms. Data refer to 2011.

	Low-input (LI)	**Mid-input (MI)**	**High-input (HI)**
Utilized Agricultural Area (ha)	125	70	67
Heads (number)	120	320	370
Stocking rate (ewes ha^{-1})	1.0	4.6	5.5
Milk production (kg $year^{-1}$)	25,000	79,655	110,000
Milk pro-capita annual production (kg ewe^{-1} $year^{-1}$)	208	249	297
Pastures - grazing area (ha)	95	52	12
Arable land – cereals and annual forage crops (ha)	30*	18	55
Concentrate feed annual consumption (t) **	1	121	204
Mineral N-fertilizing (kg ha^{-1})	0	21	45
Mineral P_2O_5-fertilizing (kg ha^{-1})	0	72	32
Irrigation	no	yes	no
Milking system	manual	mechanical	mechanical

*10% of the arable land production is used for sheep feeding; the remaining part is sold as hay and grain.
** LI produces all concentrates on farm, MI imports all concentrate feed needed, and HI imports about 86% of total requirements.

2.2. Life Cycle Assessment methodology

The methodology used to carry out the LCA study is consistent with the international standards ISO 14040-14044 (2006 a, b). The analysis was conducted using two different *functional units* (FU): 1 kg of Fat and Protein Corrected Milk (FPCM) and 1 ha of Utilized Agricultural Area (UAA). The use of these two FU allows to define and to combine productive and economic results with depletion of natural resources. All inputs and outputs referred to 1 kg of FPCM were partitioned (*impact allocation*) between milk and the other co-products on the basis of the economic value of products (economic allocation), since the “main product” (milk) of all three farms had an economic value much higher than the other co-products (wool and meat) . When co-products were obtained from the same field (e.g., triticale-barley grain and stubble), mass-based allocation was applied, since the amounts of the individual co-products were interdependent in a physical relationship.

The life cycle was assessed "from cradle to gate", including in the *system boundaries* all the input and output related to sheep milk production. All modes of transportation and distances covered within the system were also taken into account. The model system was divided into two subsystems: a) Flock, and b) Farm Impact.

a) Flock - Processes linked with the productive life of livestock.

They include all the processes related to i) the use of agricultural soil and the cultivation operations required for the production of different forages; ii) the feed input, including the consumption of forage from pastures and feed concentrates; iii) livestock operations such as shearing (once a year) and milking (performed twice a day if mechanical, once a day if manual). Each of these processes has been applied to the different categories of sheep, depending on the breeding techniques adopted by each farm, having as primary reference points the quantity and quality of sheep diet. Therefore, LCA model includes ewes and rams, each subdivided into lambs, replacement animals and adults. The ewes were grouped by physiological and productive phase (maintenance, dry and lactation).

b) Farm Impact - Processes linked with the farm structure.

They include infrastructures (milking parlour, barns, etc.), agriculture machineries and devices (tractors, ploughs, milk cooler, pumps, etc.), water and energy consumption, and consumable materials like detergents, veterinary drugs, spare parts, etc.

About 70% of life cycle inventory data were collected through 12 visits *in situ*, interviews and a specific questionnaire (farm-specific data for year 2011). The remaining data (e.g., enteric methane emissions, supplement chemical composition, etc.) were collected from available literature and databases (mostly Ecoinvent v. 2.2 developed by Swiss Centre for Life Cycle Inventories).

With the aim of assessing in a more comprehensive way the environmental performances of the case studies, three different *evaluation methods* were used: 1) IPCC, Intergovernmental Panel on Climate Change (2006), which provides estimates on greenhouse gases emitted in the life cycle of products (Carbon Footprint), expressed in kilograms of CO_2-equivalents with 100-year time horizon; 2) ReCiPe that provides a more comprehensive assessment of life cycle environmental performances (Ecological Footprint), considering 17 different categories of environmental impact, which are calculated and harmonized obtaining a single eco-indicator (Ecopoint, Pt) (Goedkoop et al., 2009); 3) Blue Virtual Water that estimates the (virtual) water content incorporated (Water Footprint) into a product, as the volume of water, expressed in l-equivalents, consumed or polluted during the entire life cycle of the product (Hoekstra et al., 2011).

The life-cycle analysis was performed under the following *simplified assumptions*: the analysis included only the amount of forage (fodder crops and pastures) consumed by flocks, after cross-checking estimated and/or measured forage production and estimated nutritional needs based on gender, age, weight, physiological stage and production level of animals. National inventories of emissions by ISPRA (2011) for CH_4 and by IPCC (2006) for N_2O were used to quantify flocks’ enteric emissions. Irrigation impacts of MI were estimated by Ecoinvent v. 2.2. Life cycle assessment calculation was made using LCA software SimaPro 7.3.3 (PRé Consultants, 2011), which contains various LCA databases.

A Monte Carlo analysis was also performed to quantify the effects of the data uncertainties on the final results and to weight the differences between the environmental performances of the three farms.

3. Results and discussion

3.1. Evaluation of the environmental performances

The main results of the environmental impact assessment are shown in Fig. 1 by farm (LI, MI, and HI), method (IPCC, ReCiPe and Blue Virtual Water), and functional unit (1 kg of FPCM and 1 ha of UAA).

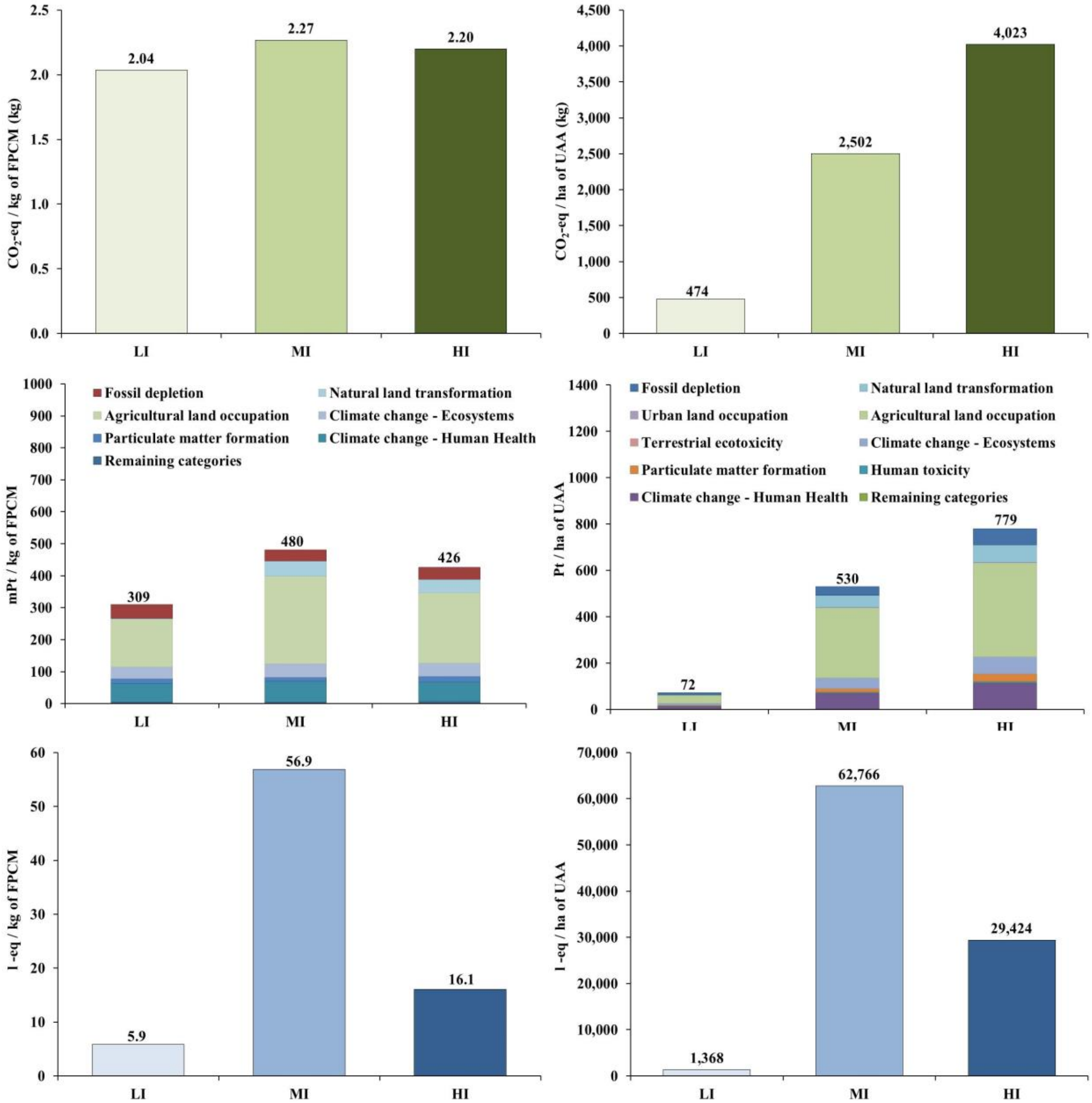

Figure 1. Main LCA results for three farms (LI, low input; MI mid input; HI, high input) using IPCC (plots at the top of the figure), ReCiPe (middle plots) and Blue Virtual Water (bottom plots) methods and two different functional units (1 kg of Fat Protein Corrected Milk, left plots, and 1 ha of Utilized Agricultural Area, right plots).

3.1.1. IPCC

The estimated life-cycle greenhouse gases (GHG) emissions of 1 kg of FPCM (plot on the left at the top of Figure 1) were slightly higher in MI. The GHG emissions per kg of FPCM from the observed production systems showed a slight variation ranging from approximately 2.0 (LI) to 2.3 (MI) kg of CO_2-eq. This result seems to contradict the findings reported in previous research studies (FAO, 2010; Hayashi et al., 2006; Michael, 2011). These authors showed that the degree of the environmental impact of farming systems referred to the quantity of product is negatively correlated with the intensity level in the inputs. The average Carbon Footprint of the three farms was equal to 2.17 kg CO_2-eq/kg FPCM and was about 61% lower than that estimated by Michael (2011) on a typical Australian dairy farm, where the Carbon Footprint was equal to 3.57 kg CO_2-eq/kg FPCM. However, the analysis of Michael (2011) was conducted on a more intensive dairy sheep farming system characterized by different sheep breed (East Friesian), and amounts of enteric methane emissions and stocking rate twice the amounts estimated in our case studies.

When the environmental impact assessment was performed using as functional unit 1 ha of UAA, the Carbon Footprint of the three studied farms showed relevant differences, indicating a strict positive relationship between the environmental impact of farms and the intensity level in the inputs (plot on the right at the top of Figure 1). The HI farm had the largest value (more than 4,000 kg of CO_2-eq), with LI and MI showing GHG emission amounts equal to about 12% and 62% of HI Carbon Footprint, respectively. Therefore, the highly extensive system of LI, with an UAA value almost twice compared to MI and HI and a stocking rate equal to 18% of HI (Table 1), was crucial in determining the environmental performance calculated per 1 ha of UAA.

3.1.2. ReCiPe

The results from the ReCiPe method assessment followed a trend similar to IPCC method for both FU (plots in the middle of Figure 1). To facilitate the interpretation of results, Fig. 1 shows only impact categories with scores higher than 10 milli-ecopoint (mPt) and 1 ecopoint (Pt) per 1 kg of FPCM (left side of the figure) and 1 ha of UAA (right side of the figure), respectively. When 1 kg of FPCM was used as functional unit, the farm with the lowest score was LI, with about 300 mPt per 1 kg of FPCM (approximately 36% and 27% less than MI and HI, respectively). When using 1 ha of UAA as functional unit, LI showed again the lowest value, with about 72 Pt per ha of UAA, which represents 14% and 10% of MI and HI impacts, respectively. It's interesting to note that the utilization of two FU led to different ranking of the three dairy farms and to a different weight of the impact categories on the life cycle assessment. Moreover, the use of 1 ha of UAA as FU determined a range of environmental impacts much higher than those assessed for 1 kg of FPCM. Agricultural land occupation was the impact category that showed the main contribution to the total environmental impact. The contribution analysis of the Agricultural Land Occupation impact category allowed to complete the description of the farming systems and to better understand the differences among the three farms in terms of input levels. The extensive land use represented more than 60% of the Agricultural Land Occupation in the LI, compared to 41% and 17% in MI and HI, respectively. On the other hand, the intensive land use was near 60% of Agricultural Land Occupation in HI, and around 35% for both MI and LI. In the latter case, the unexpected large percentage of intensive land occupation can be explained by the very limited arable land destined to forage production for sheep feeding. However, the contribution of the occupation of arable land on the total Agricultural Land Occupation was larger than 20% in MI and HI, and equal to about 1% in LI farm. Agricultural land occupation and, more generally, land use impact category are critical aspects of LCA analysis, in particular when the agricultural sector is investigated (Schmidinger and Stehfest, 2012). In this case, the impact category "land use", currently included in the most widely used LCA methods (like ReCiPe), quantifies mainly the potential impacts of land use on biodiversity, excluding other relevant land use impact categories, such as depletion of productive land area and changes in soil quality (Mattila, 2012).

3.1.3. Blue Virtual Water

Mid input farm (MI) was the only one that applied irrigation for crop production. The use of water for irrigation in MI determined, for both FU, a large Blue Virtual Water consumption compared to the other farms. The results of the Blue Virtual Water impact assessment method for 1 kg of FPCM indicate that the Water Footprint of MI was approximately 3.5 to 10 times larger than the Water Footprint observed in HI and LI, respectively (bottom plots of Figure 1). When the Water Footprint was calculated using 1 ha of UAA as functional unit, MI showed an impact value that is more than twice the HI value and about 46 times larger than the LI. In addition, the direct water consumption of MI for producing the annual total amount of FPCM was approximately 3,600 m^3, with about 65% being consumed for irrigation. The direct water consumption of HI amounted to 1,279 m^3, with an indirect consumption equal to 448 m^3. Conversely, the Blue Virtual Water consumption for LI derived mainly from indirect consumption, which represented, with 122 m^3, the 82% of the total consumption.

3.2. Contribution analysis

A detailed contribution analysis is reported in Table 2, which illustrates all processes that contributed with more than 1% to the total environmental impact of all farms for the three different evaluation methods adopted, using 1 kg of FPCM as functional unit. Similar results were also obtained for the functional unit 1 ha of UAA. In general, the analysis of the contributions of individual processes for the three farming systems showed a relevant role of enteric methane emissions, field operations (mainly tillage), electricity and production of agricultural machineries. In MI and HI, feed concentrates in the diet (in particular soy production) showed a relevant contribution, with percentages ranging from 16% for HI (IPCC method) to 30% for MI (ReCiPe method). The natural and improved pasture utilization resulted in relevant contribution only for the ReCiPe assessment method (31% in LI, 40% in MI and 34% in HI), essentially for the effect of the Agricultural Land Occupation impact category. The contribution of agrochemicals was generally low (always less than 3%), due to their very limited use in all the three farms. However, the incidence of contribution of each process varied with the evaluation method utilized. For example, the enteric methane emission is the most important impact (an overall average of 42% of total impacts) for the IPCC method, but when the estimate is performed using the ReCiPe method, the impact of the enteric methane emissions amounted on average to 11%, representing only the fifth highest-ranked impact. The complementary use of the three methods offered a multiple analysis perspective, resulting in a more comprehensive assessment of impacts. From a methodological point of view, there are many similarities between the different footprints, but each has its own peculiarities related to the uniqueness of the substances considered.

The analysis of contributions has been also useful for identifying more specific strengths and weaknesses of each dairy sheep farming system, in order to improve their environmental performances.

Enteric methane emissions represented the most important environmental impact factor for all the farms when IPCC method is used. This result is consistent with the actual knowledge about the role played by the enteric methane fermentation in ruminant livestock emissions, which are estimated to represent approximately 18% of the global anthropogenic GHG emissions (FAO, 2006). Few practical strategies can be followed for reducing enteric methane emissions of grazing animals (Hegarty et al., 2007), mainly by regulating the quantity and quality of feed consumed (Pelchen and Peters, 1998) or utilizing inhibitors of enteric fermentation (Martin et al., 2010; Nolan et al., 2010; Puchala et al., 2005; Tiemann et al., 2008; Wallace et al., 2006). However, further research studies are needed to carefully analyse the complexity of relations among breeding techniques and enteric gas emissions (e.g., methane and nitrous oxide).

Beside the crucial role of enteric methane emissions, the major contributions to the environmental impact of LI are due to land use on natural and improved pastures (i.e., field operations, ranging from 21% to 34%, for ReCiPe and Blue Virtual Water, respectively), electricity (from 8% to 16%, for ReCiPe and Blue Virtual Water, respectively), and agricultural vehicle's equipment (from 8% to 10%, again for ReCiPe and Blue Virtual Water, respectively). The power consumption of LI depended mainly on milk cooling and therefore an improvement of the environmental performance of this farm could be achieved choosing the proper size of the cooling tank and/or adopting a more efficient cooling system, possibly powered by renewable sources. In addition, LI showed a relevant contribution to the overall impact determined by tractor and other devices, such as pick-up and generator diesel (10%, 8%, and 26% for IPCC, ReCiPe and Blue Virtual Water methods, respectively). This contribution is at least double compared to the contribution observed in the other farms and it can be likely due to the use of over-dimensional and power-consuming equipment compared to the farm needs.

The contribution of field operations (tillage and sowing) to the total environmental impact of the productive cycle of 1 kg of FPCM was largely lower in MI (with values never exceeding 8%) than in the other farms, for all the adopted methods. This result could be probably due to the minimum tillage practice used by MI for sowing of pasture mixtures. However, the environmental performances of MI could be improved by reducing the purchase of feed concentrates and consequently increasing the amount of pasture and self-produced hay in the diet of flock. To achieve this result, an increase of the total surface sown with well adapted and high quality pasture mixtures may be suggested (Franca et al., 2008; Porqueddu and Maltoni, 2005). The overall high consumption of electricity suggests to introduce a farm strategy based on renewable source power supply. Finally, it may be appropriate to assess a proper sizing of the machinery stock, in relation to the needs of MI.

The contribution of concentrate feed was particularly large in MI, despite lower total annual consumption compared to HI (0.38 t ewe^{-1} versus 0.55 t ewe^{-1}). It's important to note that HI produced about 24% of its concentrate needs on-farm and had a larger annual milk yield per ewe compared to MI, which imported all concentrate. In HI, the improved pastures and annual forage crops represented the most relevant contribution to the overall environmental impact. Taking this result into account, a possible strategy to reduce the Carbon and Ecological Footprint of HI could consist in increasing the agricultural surface area utilized for permanent semi-natural pastures and finding proper pasture management strategies (i.e., deferred grazing during spring to allow self-reseeding). Moreover, improving power supply strategy could represent an effective way to enhance the HI environmental performance, as well as for the other farms.

Table 2. Percentage contribution of processes to the total environmental impact of low- (LI), mid- (MI) and high-input level (HI) farms, using three evaluation methods (ICPP, ReCiPe, and Blue Virtual Water) and 1 kg of FPCM as functional unit. The process category "Remaining processes" includes all the processes with a percentage contribution lower than 1% for all methods and farms.

	IPCC			**ReCiPe**			**Blue Virtual Water**		
Process	**LI**	**MI**	**HI**	**LI**	**MI**	**HI**	**LI**	**MI**	**HI**
Enteric methane emissions	45	46	34	14	10	8	0	0	0
Field operations (tillage and sowing)	27	8	16	21	4	8	34	1	7
Electricity, medium voltage	13	5	3	8	2	1	16	1	1
Natural pastures	1	2	0	31	24	9	0	0	0
Improved pastures	0	2	16	17	21	36	0	0	2
Concentrate feed	1	21	16	1	30	26	1	2	5
Lactating ewes (feed consumption and animal excretion)	1	1	1	0	0	0	0	0	0
Infrastructures (milking parlour, barn, etc.)	0	2	1	0	0	0	3	1	1
Irrigating (infrastructure and water consumption)	-	0	0	-	0	0	-	59	0
Tractor, production	4	2	2	3	1	1	9	0	1
Pick-up vehicle, production	1	0	0	1	0	0	2	0	0
Agricultural machinery, production	5	3	2	4	1	2	15	1	3
Transport (lorry and/or transoceanic freight ship)	0	5	4	0	1	1	0	0	1
Water consumption (milking and irrigating excluded)	0	0	0	0	0	0	18	32	70
Agrochemicals (urea, glyphosate, etc.)	-	0	3	-	0	2	-	0	1
Consumable materials (detergent, veterinary drugs, etc.)	0	0	0	0	0	0	1	0	1
Remaining processes	2	3	2	0	6	6	1	3	7

3.3. Monte Carlo analysis

Uncertainty results from the Monte Carlo simulations showed different implications depending on the functional unit and the evaluation method used (Table 3). Blue Virtual Water method showed significant differences between the three production system case studies, due to the large differences in water consumption (LI and HI water consumption were respectively 4% and 43% than MI) with relatively low uncertainty intervals (16%, 14%

and 4-5% for LI, MI and HI, respectively). No significant differences were found between the environmental performances of the three farms when using 1 kg of FPCM as functional unit and IPCC and ReCiPe evaluation methods. For both methods the uncertainty interval did not exceed 16% and the probability of a farm to have an Ecological or Carbon Footprint greater than or equal to the footprint of another farm did not exceed 22%, on average. As expected and regardless of the method used, the Monte Carlo analysis confirmed that the differences between the environmental impacts of the three farms calculated using 1 ha of UAA as functional unit were significant.

Table 3. Monte Carlo simulation output for each assessment method, functional unit and farm with low- (LI), mid- (MI), and high-input (HI) levels. Average values and uncertainty intervals for P≤0,05 (between brackets) are reported.

Farm /FU/ Method	LI		MI		HI	
	1 kg FPCM	1 ha UAA	1 kg FPCM	1 ha UAA	1 kg FPCM	1 ha UAA
IPCC (kg CO_2-eq)	2.0 ± 10%	474 ± 11%	2.3 ± 13%	2,502 ± 13%	2.2 ± 13%	4,023 ± 0%
ReCiPe (Pt)	0.3 ± 13%	72 ± 13%	0.5 ±16%	530 ± 16%	0.4 ± 15%	779 ± 15%
Blue Virtual Water (l-eq)	5.9 ± 16%	1,368 ± 16%	56.8 ±14%	62,766 ± 14 %	16.1 ± 4%	29,424 ± 5%

4. Conclusion

In this work, LCA approach was used for comparing dairy sheep production systems and for identifying the hotspots to improve their environmental performances. The LCA conducted with two different functional units (1 kg of Fat Protein Corrected Milk and 1 ha of Utilized Agricultural Area) and three different assessment methods (IPCC, ReCiPe, and Blue Virtual Water) led to a more objective evaluation of the environmental performances of three case studies, taking into account both the economic dimension and the environmental role of dairy farming systems.

The environmental performances of the studied farming systems were similar when using 1 kg of FPCM as functional unit, regardless of the assessment method used. On the other hand, the environmental impacts were significantly different when the assessment was based on the functional unit 1 ha of UAA. Moreover, the contribution of each process to the environmental performances of dairy sheep systems depended largely on the method used for the evaluation. However, this study shows the relevant role played by enteric methane emissions, tillage, electricity and machineries in the overall environmental performances. Feed concentrates in the diet (in particular soy production) showed a relevant contribution in MI and HI. The natural and improved pastures utilization resulted in relevant contribution only for the ReCiPe assessment method. The contribution of agrochemicals was generally low, due to their very limited use in all the farms.

Certainly, LCA represents a valid approach for developing an eco-design strategy at farm level aimed to exploit the multifunctionality of extensive Mediterranean livestock systems. Further studies and knowledge are needed to overtake the limits of the LCA methodology when applied to agriculture sector. In particular, a certain degree of weakness derives from the lack of both site-specific datasets on Mediterranean farming systems and appropriate methods to assess comprehensively the “land use” impact category.

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Construction cost of plant compounds provides a physical relationship for co-product allocation in life cycle assessment

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ABSTRACT

Although, according to the ISO 14044 standard, economic allocation should be the solution of last resort, it is frequently used in LCA studies of agri-food systems because of the inadequacy of solutions preferred by ISO. We calculated allocation factors for a range of plant co-products used as feed ingredients according to three methods: energy allocation (En), economic allocation (Ec) and allocation based on plant physiological construction cost of plant compounds (Cc). Compared to En and Ec, Cc yields higher allocation factors for protein-rich co-products and lower factors for lipids. Whereas the difference between En and Cc is modest (up to 5 percentage points), the difference between Ec and Cc is more variable and can be large (up to 18 percentage points). For plant co-products, Cc is an attractive option, as it is based on the physiological mechanisms involved in plant growth rather than on a common property of the co-products.

Keywords: economic allocation, energy allocation, plant co-products, (biophysical mechanism)

1. Introduction

For a long time, the topic of allocation has been the subject of methodological debate in Life Cycle Assessment (LCA) studies. The International Organisation for Standardisation (ISO) 14044 standard (ISO 2006) for LCA defines allocation as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems". Allocation is about attributing the relative shares of responsibility for environmental impacts to several co-products (Suh et al. 2010; Pelletier and Tyedmers 2011).

The ISO 14044 standard defines the following procedure for allocation:

1. Wherever possible, allocation should be avoided by
   a) dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or
   b) expanding the product system to include the additional functions related to the co-products.
2. Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them.
3. Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products (ISO 2006).

Agri-food systems contain many multiple output processes, e.g. the transformation of soybeans yielding meal and oil, or the production of milk, calves and cull cows by a herd of dairy cows. How to best allocate impacts to co-products has been the subject of much debate (Suh et al. 2010; Ardente and Cellura 2011; Pelletier and Tyedmers, 2011). Although, according to ISO guidelines, economic allocation should be the solution of last resort, it is frequently used in LCA studies of agri-food systems because of the inadequacy of the solutions that should be preferred according to the ISO decision hierarchy.

Avoiding allocation by dividing the unit process often is not possible, as for instance in the case of the transformation of soybeans in meal and oil. Avoiding allocation by expanding the system is feasible for those co-products that can be produced by other processes. However, for agricultural products this approach is often problematic. For instance, in the case of a dairy herd producing milk, cull cows and calves, we may want to identify a process that yields a product similar to cull cows, i.e. animals that can be transformed into beef. One can identify a suckler cow herd as an alternative producer of animals that can yield beef. However, on close examination this is not satisfactory, as beef from a suckler system is of a different (better) quality than beef from culled dairy cows. Finding an alternative production process for cow milk is even more difficult. One might turn to processes producing goat milk or soy "milk", but these are obviously very different products. As a consequence, as system expansion applied to LCAs of agricultural products often fails to identify alternative processes producing the same products; it generally resorts to choosing alternative processes yielding products used for the same purpose, i.e. insuring functional equivalence. LCA studies will for instance consider that suckler cow beef can be considered equivalent to dairy cow beef (Nguyen et al. 2013), or that pork is equivalent to beef (Thomassen et al. 2008a). Considering that low and high quality beef are equivalent, or that pork is equivalent to beef may, for a variety of reasons, not only seem quite far-fetched to most people, it also introduces major arbitrariness and additional uncertainty in the LCA study. Last but not least, the system expansion approach causes the LCA model to get larger and more complicated, requiring the collection of more data (Curran 2013).

When allocation can not be avoided either by dividing the unit process or by expanding the product system, then the next best option, according to ISO 14044, is to allocate the inputs and outputs between the products in a way that reflects the underlying physical relationships between the products. In their eloquent but in parts hard-to-understand paper “An ecological economic critique of the use of market information in life cycle assessment research” Pelletier and Tyedmers (2011) critique the use of economic allocation and plead for the use of physical or biophysical properties of products in allocation. They admit however that biophysical allocation is harder to justify when it is necessary to allocate burdens between co-products with divergent functions, for example when soy processing co-products are used as meal for animal nutrition and as oil for biodiesel. In such a case, finding a common physical or biophysical property adequate for both functions is difficult. Furthermore, even when one can find such a property, one may wonder to what extent the identification of a common physical or chemical property of the co-products can be equated to the existence of an underlying physical relationship between the co-products as mentioned in the ISO 14044 standard. Although this seems to have been accepted as a common interpretation of the ISO standard in many published papers on the subject, it remains quite a puzzling assumption to the authors of this paper.

So, since both avoiding allocation as well as allocating according to a physical relationship rarely seem to work satisfactorily for agri-food multi-output systems, it should not come as a surprise that many LCA studies of such systems end up allocating inputs and outputs to co-products in proportion to their economic value (Ardente and Cellura 2011; Brankatschk and Finkbeiner 2014). For many authors, economic allocation is attractive as it is the value created that causes the process. However, economic allocation is not without difficulties. It is based on the one hand on the relative masses of the co-products produced and on the other hand on the relative economic values of these co-products. While the former are generally quite stable, the latter tend to fluctuate with time and place in response to changes in supply, demand, regulation, subsidies, culture (Pelletier and Tyedmers 2011) and technological developments. As a result allocation factors will fluctuate, affecting the credibility of the results of the LCA. This problem can be reduced by averaging product prices over several years which will smooth short-term variations

In this paper we want to contribute to this debate. We first present an inventory of the use of methods to attribute impacts to plant co-products used as animal feed in recent LCA studies to quantify the relative importance of these methods as they are actually used in studies published in peer-reviewed publications. Secondly we propose a new way of allocating impacts to plant co-products according to the underlying physical relationships between the co-products. We compare this method to two current allocation methods.

2. Methods

2.1. Attribution of impacts to plant co-product in LCA studies

We found 24 LCA studies from peer-reviewed scientific journals and scientific reports for the 2000-2013 period that attributed impacts to plant co-products used as animal feed. These studies used either system expansion or allocation according to co-product mass, energy content, nitrogen content or economic value, or several of these methods. This allowed us to establish the relative share of co-product handling methods used in “real life”, so to say.

2.2. Comparison of three allocation methods

This case study focused on plant co-products used as ingredients for animal feed. We compared three methods for the allocation of impacts to plant co-products: energy allocation, economic allocation and the new allocation method (presented in this paper) according to the construction cost of plant compounds.

Energy allocation is often used to allocate impacts when one or more of the co-products are used as biofuels (ADEME 2010). Energy allocation is generally based on the lower heating value (LHV) of the co-products. For some plant co-products not associated with biofuel production (e.g. wheat middlings) LHV data are not available in the literature. Furthermore, in many biofuel studies, LHV values used for the various co-products are taken from several literature sources, introducing methodological heterogeneity for these data. In order to obtain a coherent set of LHV values for the co-products we studied we estimated their LHV values from their gross energy values taken according to INRA (2004; 2007). Gross energy is defined as the heat produced during the complete combustion of an organic component in a calorimeter in the presence of oxygen (INRA 2007). Given this definition, a gross energy value therefore corresponds to a higher heating value (HHV). It has been observed that the LHV of co-products used as animal feed is 6 to 7% lower that their HHV (Hofstrand 2008). We assumed a 6% difference to estimate LHV from gross energy values, as we thus obtained LHV values that were closest to those found in literature sources.

For economic allocation we used average annual prices for the 2005-2009 period. These were world market prices for oils and meals from soybean, rapeseed, sunflower, linseed, oil palm, and for corn oil (ISTA 2009; 2011); for wheat and maize co-products other than corn oil these were prices for the United States of America (USDA 2012), for sugar beet co-products these were prices for France (AGRI C5 2013; I. Bouvarel pers. Comm. 2013). These prices were used to calculate so-called “Olympic averages” i.e. for each co-product we took the five annual average values for the 2005-2009 period, eliminated the highest and lowest values, and averaged the values for the remaining three years.

We propose a new allocation method for plant co-products based on physiological mechanisms involved in plant growth rather than on physical or chemical characteristics of the co-products. Plant biomass results from photosynthesis. Plants convert glucose, the prime product of photosynthesis, into other plant compounds such as carbohydrates, proteins, lipids, lignin and organic acids. These groups of compounds differ in their construction costs, i.e. g of glucose/g of compound (Poorter, 1994). Plant co-products (e.g. soybean meal and soybean oil) differ in their contents of these compounds and consequently have different construction costs. We propose biomass construction cost as a characteristic for physical allocation.

Construction cost was estimated according to Poorter (1994) using the following equation:

$$CC = (-1.041 + 5.077 * C_{om}) * (1\text{-}M) + (5.325 * Norg)$$

Where:
CC: the total cost to produce one gram of plant biomass (g glucose/g dry weight)
C_{om}: the carbon content of the biomass (g/g dry matter)
M: the mineral content of the biomass (g/g dry matter)
N_{org}: the organic nitrogen content of the biomass (g/g dry matter)

The carbon content of the biomass was estimated according to Vertregt and Penning de Vries (1987) using the following equation (ignoring organic anions):

$$C_{om} = 0{,}444 * \text{carbohydrates} + 0{,}535 * \text{protein} + 0{,}774 * \text{lipids} + 0{,}667 * \text{lignin}$$

All components as well as C_{om} are expressed as g/g dry matter (DM). Carbohydrate, protein, lipids, lignin and mineral contents of co-products were taken from INRA (2004; 2007), from the Feedipedia (2013) website and from ANSES (2013).

The calculation of allocation coefficients for the three methods requires extraction rates (the proportion of the processed product obtained from the parent product). Extraction rates for soybean, rapeseed and sugar beet were from Jungbluth et al. (2007), for sunflower from FAO (2002), for linseed from unpublished INRA data, for oil palm from Schmidt (2007), for wheat and maize from Würdinger et al. (2002).

3. Results

3.1. Attribution of impacts to plant co-product in LCA studies

In the studies surveyed, economic allocation was used in 19 out of 24 cases (Table 1). Energy allocation was used in three studies, system expansion and mass allocation were used in two studies, while nitrogen content was used in one study. This illustrates the popularity of economic allocation among scientists dealing with LCAs involving plant co-products, in spite of ISO 14044 recommending economic allocation as a last resort.

Table 1. The use of methods to attribute impacts to co-products used as animal feed in LCA studies published in peer-reviewed journals in the 2000 – 2013 period.

Study reference	Main product studied	System expansion	Allocation method			
			Mass	Energy (gross energy)	Nitrogen content	Economic
Cederberg and Mattsson 2000	Milk					X
Haas et al. 2001	Milk					X
Cederberg and Stadig 2003	Milk					X
Casey and Holden 2005	Milk					X
Basset-Mens and van der Werf 2005	Pig					X
Casey and Holden 2006	Beef cattle					X
Mollenhorst et al. 2006	Eggs					X
Williams et al. 2006	Milk, meat, eggs					X
Katajajuuri 2008	Chicken					X
Pelletier 2008	Chicken			X		
Thomassen et al. 2008a	Milk	X	X			X
Thomassen et al. 2008b	Milk					X
Van der Werf et al. 2009	Milk					X
Pelletier et al. 2010a	Pig			X		
Pelletier et al. 2010b	Beef cattle			X		
Nguyen et al. 2010	Beef cattle	X				
Gerber et al. 2010	Milk					X
Cederberg et al. 2013	Milk, meat, eggs					X
O'Brien et al. 2012	Milk		X			X
Weiss and Leip 2012	Milk, meat, eggs				X	
Nguyen et al. 2012a	Chicken					X
Nguyen et al. 2012b	Beef cattle					X
Ripoll-Bosch et al. 2013	Sheep					X
Nguyen et al. 2013	Milk					X
Total	**24**	**2**	**2**	**3**	**1**	**19**

3.2. Lower heating value, average price and construction cost for the co-products

The lower heating values (in MJ/kg DM) that we calculated from gross energy values were 36.9 for oils and fat, for oilseed meals they ranged from 18.1 for rapeseed meal to 18.5 for soybean meal; for the other co-products they ranged from 14.5 for beet molasses to 18.9 for palm kernel (Table 2). Prices for oils (in $/t gross product) ranged from 626 for palm oil to 1028 for linseed oil; for oilseed meals they ranged from 178 for sunflower meal to 321 for soybean meal; for the other co-products they ranged from 69 for corn gluten feed to 585 (Euros) for beet sugar. Construction costs (in glucose/kg DM) were 2.89 for oils and fat, for oilseed meals they ranged from 1.54 for sunflower meal to 1.67 for soybean meal; for the other co-products they ranged from 0.99 for beet molasses to 2.20 for palm kernel.

Table 2. Lower Heating Value (LHV), Olympic average price for 2005-2009 and construction cost of plant co-products used for animal feed

Co-product	LHV (MJ/kg DM[1])	Price 05-09 ($/t gross product[2])	Construction cost (kg glucose/kg DM)
Soybean oil	36.9	694	2.89
Soybean meal	18.5	321	1.67
Ratio oil/meal	1.99	2.16	1.73
Rapeseed oil	36.9	874	2.89
Rapeseed meal	18.1	184	1.59
Ratio oil/meal	2.04	4.75	1.82
Sunflower oil	36.9	851	2.89
Sunflower meal	18.2	178	1.54
Ratio oil/meal	2.03	4.78	1.88
Linseed oil	36.9	1028	2.89
Linseed meal	18.2	293	1.58
Ratio oil/meal	2.03	3.51	1.83
Refined palm oil	36.9	626	2.89
Palm kernel	18.9	-	2.20
Fodder fat	36.9	-	2.89
Wheat flour	15.4	313	1.42
Wheat bran	17.8	82	1.34
Wheat middlings	17.9	82	1.39
Feed grade wheat flour	17.8	82	1.40
Maize starch	16.5	319	1.22
Corn gluten feed	17.6	69	1.32
Corn oil	36.9	760	2.89
Corn gluten meal	21.7	370	2.07
Beet sugar	15.7	585[3]	1.21
Pressed sugar beet pulp	16.0	134[3]	1.13
Beet molasses	14.5	160[3]	0.99

[1] Dry Matter
[2] Product at its reference humidity content
[3]€/t gross product

For LHV oil/meal ratios for the four oilseeds (soybean, rapeseed, sunflower and linseed) varied little and ranged from 1.99 for soybean to 2.04 for sunflower (Table 2). For prices the ratio varied widely, ranging from 2.16 for soybean to 4.78 for sunflower. For construction cost the ratio varied moderately, ranging from 1.73 for soybean to 1.88 for sunflower. For soybean oil/meal ratios varied least across the three methods (1.73 to 2.16), whereas for the other three oilseeds these ratios varied much more across methods: rapeseed: 1.82 to 4.75, sunflower 1.88 to 4.78 and linseed 1.83 to 3.51.

Construction cost was strongly correlated with LHV, price was correlated with both LHV and construction cost, but the correlation was weaker (Table 3).

Table 3. Correlation coefficient (r) between lower heating value (LHV), Olympic average price for 2005-2009 and construction cost of plant co-products used for animal feed

	LHV	Price 05-09	Construction cost
LHV	1.0	0.697	0.987
Price 05-09		1.0	0.722
Construction cost			1.0

Table 4. Allocation factors according to energy content, economic value and construction cost for plant co-products used for animal feed

Raw material	Energy content	Economic value	Construction cost
Soybean oil	35.3	34.2	32.1
Soybean meal	64.7	65.8	67.9
Rapeseed oil	60.0	75.6	57.3
Rapeseed meal	40.0	24.4	42.7
Sunflower oil	63.0	78.1	61.2
Sunflower meal	37.0	21.9	38.8
Linseed oil	46.9	57.5	44.3
Linseed meal	53.1	42.5	55.7
Refined palm oil	85.0	81.8	80.5
Palm kernel	11.5	15.0	16.1
Fodder fat	3.6	3.2	3.4
Wheat flour	76.7	93.6	79.8
Wheat bran	13.6	3.8	11.6
Wheat middlings	4.0	1.1	3.5
Feed grade wheat flour	5.7	1.6	5.1
Maize starch	62.5	79.2	61.0
Corn gluten feed	24.9	6.4	24.6
Corn oil	6.4	7.5	6.5
Corn gluten meal	6.3	6.9	7.9
Beet sugar	67.3	72.8	69.4
Pressed sugar beet pulp	22.8	23.0	21.6
Beet molasses	9.9	4.2	9.0

3.3. Allocation factors for the co-products

For soybean, energy (En) and economic (Ec) allocation factors were close (Table 4). For the other oilseeds Ec allocation had higher factors for oil than En allocation, but for oil palm Ec allocation had a lower factor for oil than En allocation. For co-products from wheat, maize and sugar beet Ec allocation had higher factors for the main product (flour, starch and sugar, respectively) than En allocation.

Relative to En allocation, construction cost (Cc) allocation had a lower factor for oil for the four oilseeds as well as for oil palm. For wheat co-products, Cc had a higher allocation factor for flour and a lower allocation factor for bran than En. For maize and sugar beet co-products En and Cc allocation factors were quite close.

4. Discussion

The survey on the application of the different approaches for the attribution of impacts to plant co-products yielded quite sobering results, as it revealed a large gap between the normative propositions on co-product handling in the ISO guidelines and the actual practice by scientists publishing LCA studies. Clearly neither avoiding allocation nor allocation according to a physical relationship proved satisfactory for plant co-products to most LCA practitioners. There may therefore be interest among LCA scientists for a new approach to allocation of plant products.

Allocation according to the underlying physical relationships between the co-products, as recommended in the ISO 14044 standard, is generally done by identifying a common physical chemical property of the co-products such as mass or energy content (Pelletier and Tyedmers 2011). However, the identification of a common property such as mass or energy content does not really establish or reflect a common relationship between co-products. Any two products in the world have a mass and an energy content, but this does not mean they are related. The relationship between co-products is their common origin, so a physical allocation method should not be based on the physical properties of the co-products, but rather on the mechanism reflecting their common origin, as this is the basis of their relationship.

The new method proposed in this paper corresponds in our view to a more appropriate interpretation of the ISO 14044 standard, as it is not based simply on a physical property of the co-products, but on their construction cost, i.e. the common plant physiological mechanism at the origin of the different compounds that make up plant biomass. Although construction cost is strongly correlated with LHV, it is not identical. Whereas for LHV oil/meal ratios for the four oilseeds analyzed in this paper are around 2, these ratios vary between 1.73 and 1.88 for construction cost. As a result, relative to allocation factors based on energy content, allocation factors based on construction costs are higher for meals than for oils

5. Conclusion

Although, according to the ISO 14044 standard, economic allocation should be the solution of last resort, it is frequently used in LCA studies of agri-food systems because of the inadequacy of solutions preferred by ISO. Allocation according to the underlying physical relationships between the co-products is generally done by identifying a common physical chemical property of the co-products, but this does not really reflect a relationship between the co-products. We propose construction cost of plant compounds making up the co-products as an allocation criterion which is truly based on a relationship between the co-products.

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Defining a nutritionally healthy, environmentally friendly, and culturally acceptable Low Lands Diet

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ABSTRACT

Our study quantifies the historical Dutch diet of 80 years ago, based on a cultural history research. We calculate the greenhouse gas emissions (GHGE) and land use (LU) of this diet, using actual LCA data for 206 most consumed products, and the health score, based on ten nutritional characteristics. In order to meet the current requirements, we optimize this diet for adult males using linear programming. We compare the diet with the present Dutch, Mediterranean and New Nordic Diet. An optimized Low Lands Diet has the same healthy nutritional characteristics as the Mediterranean and the New Nordic Diet, and results in a lower environmental impact. The diet consists mainly of products with a high nutrient density. An adaptation of this historical diet, which fits more into the eating habits, climate and agricultural tradition of the Low Lands, is assumed easier to achieve than a transition into a foreign European diet.

Keywords: Greenhouse gas emissions, historical Dutch diet, New Nordic Diet, linear programming, health impact

1. Introduction

The present Dutch diet is not in line with the dietary guidelines (Health Council 2006; WHO 2003). Regarding the health impact, the diet has an unfavorable fatty acid composition, which increases the risk of cardiovascular diseases. Besides, only 5% of the Dutch population adheres to the recommended fatty acid composition through fish consumption. Insufficient consumption of fruit and vegetables increases the risk of coronary heart diseases, stroke and some forms of cancer (lung, breast and stomach cancer) (Van Kreijl et al. 2006). The increased intake of energy-dense foods that are high in fat has resulted in a worldwide doubling of obesity since 1980. In The Netherlands 30% in the population aged 4 years and older are moderately overweight. Another 10% has severe overweight (obesity) (CBS, 2012). Overweight and obesity are major risk factors for a number of chronic diseases, including diabetes, cardiovascular diseases, and cancer. The unfavorable dietary composition contributes approximately twice as much to the total mortality as overweight: 10% versus 5% of the annual deaths in the Netherlands (Van Kreijl et al. 2006).

Current trends in food production and consumption are considered unsustainable. For example, approximately one-third of human influence on greenhouse gas emissions (GHGE) and land use (LU) is related to our diet and the food chain (Vringer et al. 2010). The average Dutch diet is less sustainable, in terms of GHGE and LU, than eating according to the Dutch dietary guidelines. Further improvements are within reach, we conclude in a recent paper (Van Dooren et al. 2014a).

To reach these improvements, policy makers are exploring the possibilities to develop guidelines for healthy diets that are also low in environmental impacts (Health Council 2011). These guidelines are still qualitative and quite general. The aim of this paper is to define more in detail a nutritionally healthy, environmentally friendly and culturally acceptable diet for the Low Lands. In theory it is possible to define an optimal diet, according to criteria both for health and environmental impact (Van Dooren et al. 2014b). However such optimized diet may have difficulties being broadly accepted by consumers. Therefore, we seek dietary solutions that fit into the historical and cultural context, and climate of the Netherlands and Flanders, called the Low Lands Diet[1]. We build on the experience gained by scientists who recently explored and defined the Mediterranean diet (MD) (Bach-Faig et al. 2011; Fidanza and Alberti 2005), and the 'New Nordic Diet' (NND) (Bere and Brug 2009; Bere and Brug 2010; Hahnemann 2010; Mithril et al. 2013; Mithril et al. 2012; Uusitupa et al. 2013). Our hypothesis is that through clever combinations we can define an optimal diet for the Low Lands of Europe, comparable to the MD and NND.

[1] *Low Lands refer to the Rhine-Meuse delta, mainly characterized by fertile flat land at North Sea level, a temperate climate, and a sober, tolerant, former Calvinist, Dutch speaking culture.*

2. Methods: approach

In order to substantiate our hypothesis and reach our aim, we first describe our method to quantify the nutritional health (section 2.1) and environmental impact of different diets (2.2). To calculate the environmental impact, we need the GHGE and LU data of food products. Reliable data are sourced; using actual Life Cycle Assessments for 206 most consumed Dutch products (2.3). We apply this calculation methods to diets found in literature (2.4).

2.1. Calculating the Health Score

The health benefits and impacts of diets are highly complex and under continual debate and not easy to quantify. Insight in and impact of different indicators are relative and change overtime. Fortunately, different health organizations such as WHO (2003), World Cancer Research Fund, RIVM, and the Dutch Health Council (2006) have been using more or less the same indicators. The existing concept of the US Healthy Eating Index (Kennedy et al. 1995) to quantify overall diet quality is useful, but not directly applicable to Europe due to differences in cultural habits (serving sizes) and nutritional guidelines. Therefore, we developed a related score relevant to the European context. This Health Score is described elsewhere (Van Dooren et al. 2014a). The Dutch Health Council (2006) has translated and quantified the ten indicators towards a recommended intake in the Dutch population. These indicators reflect preventive factors for different food-related diseases, such as obesity, heart disease and cancer. We calculate the scores of diets, based on the ten indicators, with the following formula: Eq.1.

Health Score = (g vegetables/200g + g fruits/200g + g fiber/30g + g fish/37g + 30 en%/en% total fat + 10 en%/en% saturated fat + 1 en%/en% trans-fat + 10 en%/en% free sugars + 6 g/g salt + 2500 kcal/kcal energy)/10 **Eq. 1** (Van Dooren et al. 2014a)

2.2. Calculating the Sustainability Score

In 2010 the FAO agreed upon a definition for sustainable diets (FAO 2010): *Sustainable Diets are those diets with low environmental impacts which contribute to food and nutrition security and to healthy life for present and future generations.* 'Low environmental impacts' –as part of the definition of sustainable diets- need to be quantified using different parameters. Energy use and GHGE can be considered good proxies for this total environmental impact (Dutilh and Kramer 2000). LU and land use change are good proxies for loss of biodiversity (Pereira et al. 2010). In the score, we therefore used LU and GHGE as indicators to quantify -in relative terms- the environmental impact of the diets, because together they cover at least the top 4 of the impacts identified by Rockström et al. (2009). Our Sustainability Score is defined in an earlier study (Van Dooren et al. 2014a). We used the 2020 European Commission goal of a 20% GHGE reduction in the food chain (compared to 1990) as a reference value, although it is a political, arbitrary choice. The GHGE of the Dutch diet in the 1990s was 4.09 kg CO_2eq/day; the 2020 goal is 20% lower or 3.27 kg CO_2eq/day. This level was allocated a score of 100.

For LU we have no single reference value. The present LU in the Netherlands, according to the WWF (2012), is 3.20 global hectares. Publications about the ecological footprint (WWF 2012) suggest that the worldwide available biocapacity calculated in LU is 1.78 global hectares, which is 44% below the present use. We used as a reference a 44% reduction in the food chain applied for LU. From 5.34 m^2*year/day (the LU of the 1990s Dutch diet) to 2.97 m^2*year/day is a 44% reduction. This value was indicated as 100. The Sustainability Score was defined as the average of the GHG and LU score per diet. The score was calculated with the following formula: Eq.2.

Sustainability Score = (3.27 kg/ kg CO_2eq GHG + 2.97 m^2*year/ m^2*year LU)*100 /2
Eq. 2 (Van Dooren et al. 2014a)

2.3. Life Cycle Assessment of food products

Our calculation of GHGE and LU for the most consumed products in the diets is based on Life Cycle Assessment (BSI 2008; JRC 2010). The LCA methodology for agricultural products we use is described in 'The Agri-footprint method; Methodological LCA framework, assumptions and applied data, Version 1.0' (Blonk et al. 2011). The life cycle boundary is from raw materials acquisition and natural resources to final disposal, including food waste, and some estimate of energy use for cooking and preparation at home. In LCAs of agricultural products, the main contributors to the end score are GHGE, LU, and fossil energy use (Sevenster et al. 2010). Two of these are covered. In the scope of this study, it was not possible to carry out an extensive assessment to determine standard deviations for the parameters. LCA experts assume a general uncertainty of 10% to 20% in the results (Blonk et al. 2011).

2.4. Literature research

Using literature research, we define European examples of diets which are mentioned as both nutritionally healthy and low in environmental impact. We focus on recent publications exploring the Mediterranean and New Nordic Diets. In order to compare these diets with the Low Lands Diet, we quantify also the historical Dutch diet of about 80 years ago (1900-1940), based on a cultural history research (Jobse-Van Putten 1995; Van Otterloo 1990).

3. Results

3.1. Defining the Mediterranean Diet (MD)

The MD has been the subject of many studies (Trichopoulou et al. 2005) and we have previously studied this diet (Van Dooren et al. 2014a). The MD is characterized by a high intake of vegetables, pulses, fruits, and cereals (in the past largely unrefined); a moderate to high intake of fish; a low intake of saturated lipids but high intake of unsaturated lipids, particularly olive oil; a low to moderate intake of dairy products, mostly cheese and yogurt; a low intake of meat; and a modest intake of alcohol, mostly as wine (Willett et al. 1995). The best quantitatively defined description of this historical diet is probably published by (Fidanza and Alberti 2005).

Harvard Medical School further translated the dietary pattern into a more Western, culturally acceptable, diet with concrete recommendations (Willett 2001) and Willett published together with Oldways in 2009 the Mediterranean Diet Pyramid (www.oldwayspt.org). A consensus meeting recently updated the Mediterranean diet pyramid (Bach-Faig et al. 2011). Buchner et al. (2010) published the Double Pyramid and compared the ecological footprint of foods with their health-related position in the pyramid. They concluded that foods that are recommended for health reasons generally have lower environmental impacts as well. In contrast, foods with lower recommendations are those with a higher environmental impact. In an earlier climate diet study the Willett diet was also evaluated (Stehfest et al. 2009). In Table 1 we compare three ways to quantify the MD. The last column has been used for this study.

Table 1. Quantification of the Mediterranean Diet.

(Fidanza and Alberti 2005) *(% of energy)*	**(Bach-Faig et al. 2011)** *(servings (s))*	**(Van Dooren et al. 2014a)** *(grams per day)*
cereals (48%–52% of energy)	1-2 s pref. wholegrain every meal	210 g wholegrain bread
		100 g grain products (pasta)
extra virgin olive oil (14.5%–16.6%)	olive oil every meal	45 g vegetable oils
vegetables (5%–7%)	>= 2 s every meal	300 g mainly fresh vegetables
	potatoes <= 3s weekly	25 g potatoes
pulses (4.4%–6.6%)	>= 2 s weekly	75 g pulses
fruit (2.0%–2.6%)	1-2 s every meal	250 g fruits
fish (1.6%–2.0%)	>= 2 s fish/seafood weekly	2 times a week*
red wine (4.2%–6.0%)	wine in moderation	150 ml red wine (1 glass)
	1-2 s olives, nuts, seeds daily	
	herbs, spices, garlic, onions daily	
meat (2.6%–4.0%)	< 2 s red meat weekly	once a week beef or pork*
	<= 1 s processed meat weekly	
	2 s white meat weekly	once a week chicken*
milk and dairy products (1.3%–1.8%)	2 s dairy daily (pref. low fat)	300 ml milk or dairy products
		15 g cheese
eggs (0.8%–1.4%)	2-4 s weekly	3 eggs a week*
animal fats (1.0%–2.0%)		
	<= 2 s sweets weekly	200 kcal non-basic products
		**100 g animal products total*

Epidemiological evidence suggests that a traditional MD may be beneficial to health (Keys 1980). The term ‘Mediterranean Diet’ originated from the 1960s when studies suggested that people living in Crete and South Italy were having a lower incidence of coronary heart disease (Keys 1970). The Health Score of the MD is 122, which is higher than the recommended guidelines (Table 3), and the Sustainability Score for males (86) is below the guidelines. The question is: Is this healthy, cultural accepted Southern European diet transferrable to Northern European areas?

3.2. Defining the New Nordic Diet (NND)

We find an answer as we look into the Nordic Diet. Although the Mediterranean cuisine is also popular in Western European countries, food patterns differ significantly throughout different countries. We came across a Scandinavian variant of the Mediterranean diet, the NND. This Northern European diet consists of many local Scandinavian traditional products such as fish, berries, cabbage, rapeseed oil, rye, oats and game (Hahnemann 2010). Dietary components with substantial evidence of health-promoting properties that are part of the Nordic Nutritional Recommendations 2012 (Norden 2014) are naturally included in the NND; e.g. fruits, vegetables, potatoes, whole grains, nuts, fish and shellfish (Mithril et al. 2012). The diet is quantified in Table 2. The ‘New’ diet is also based on long local traditions; in the Middle Ages foods such as fresh herbs, legumes, cabbage and root vegetables played a major role in the Nordic Diet, but their use has decreased significantly over recent decades (Mithril et al. 2012).

Table 2. Quantification of the New Nordic Diet according to Uusitupa and Mithril.

Healthy Nordic Diet (Uusitupa et al. 2013)	**New Nordic Diet (Mithril et al. 2012)**
500 g/week wholegrain pasta and rice (25 en% whole grains)	
210 g bread/d (6 slices)	275 g wholegrain bread and cereals
>50% as rye, barley or oat	
150-200 g berries	75 g berries
175 g fruits	250 g fruits
175 g vegetables	150 g root vegetables
	30 g cabbages, 240 g other vegetables
	45 g legumes
	175 g potatoes
rapeseed or sunflower oil (2/3 unsaturated fats)	15 g rapeseed oil
soy oil based margarines	10 g butter
nuts and seeds unsalted	30 g nuts and seeds
	6 g wild plants, mushrooms, herbs
2 portions/d low-fat milk and cheese	450 g low-fat milk (+50 g other dairy)
	25 g cheese
3 portions fish/week (2 fatty)	43 g fish and shellfish (5g seaweed)
white meat, poultry, game	100 g meat (4 g game)
	25 g eggs (1/2)
50 g apple juice	
	75 g extras
	15 g sugar
	1000g coffee, tea, water (600g)

The NND can be described by overall guidelines: (a) more calories from plant foods and fewer from meat; b) more foods from the sea and lakes; and c) more foods from the wild countryside (Mithril et al. 2012). According to studies by Bere and Brug (2009, 2010), this diet has a positive health impact, and a low environmental impact, similar to the MD. Bere and Brug advise moderation of beverages such as coffee, soft drinks, fruit juice and alcohol, and fatty and sweet extras in between meals in order to increase the scores.

Our calculation results in a higher Health Score of 134 for the NND and results in the same Sustainability Score of 86 (see Table 3), compared to the MD. There is some research that confirms the health impact of the NND. Uusitupa et al. (2013) for instance investigated the effects of an isocaloric healthy Nordic diet on insulin sensitivity, lipid profile, blood pressure and inflammatory markers in people with metabolic syndrome. Uusitupa et al. concluded that a healthy NND improved lipid profile and had a beneficial effect on low-grade inflammation. In 2009, OPUS was launched as a comprehensive research project for optimal well-being, development and health of Danish children through a healthy NND. The preliminary results of the scientific research on the correlation between weight loss and the NND show that eating Nordic food is an effective way to curb obesity in the Danish population (Thorson et al. 2013).

Mithril et al. (2012) concluded that the principles and guidelines of the NND could be applied in any region. In the next section we will look if an application to the Low Lands is reasonable.

3.3. Defining the Low Lands Diet (LLD)

Although the Scandinavian culture stands closer to the Low Lands than the Mediterranean, it still differs significantly. The challenge is to define and investigate a LLD, with comparable qualities to the MD or NND. A similar health effect as the NND can be expected in the Low Lands with a traditionally, predominantly plant-based diet (semi-vegetarian), also with lots of fresh and regional vegetables, fruits and whole grains (bread, pancakes, porridge), a local vegetable oil, supplemented with limited amounts of fish, eggs, meat, and milk.

The Dutch diet from the beginning of the 20th century (1900-1940) is well described by Jobse-Van Putte and fit into this description. Research was done by interviewing older people from different areas and analyzing official documents. There are no quantitative diet surveys available prior to the 1960s. Other sources confirm the findings from Jobse-Van Putte (1995) (Knibbe 2001; Van Otterloo 1990). The following issues are typical for the LLD:

- People constrain themselves, especially to price of food. It should be cheap, fast and easy.
- There is an international, cosmopolitan mind-set, because the area is in between three cultural circles: Roman, Anglo-Saxon and German/Scandinavian. Culture, as well language, and foods are borrowed from different cultures.
- The Protestant-Calvinistic religion has resulted in an ethical relation towards food and abstinence, temperance, and pleasure, and an attitude towards food as a way to survive.
- The democratic, liberal and caring government is comparable to the Scandinavian countries, with comparable values towards food as a necessity, combined with a natural, balanced, and nutritious proposition.
- Due to an open trading economy, there is a lot of tolerance towards food habits of immigrants (Chinese, Indonesian, Jewish, Italian, Greek, Turkish and Moroccan) and an early adaption of coffee, tea, rice and spices (Albala 2003; Jobse-Van Putten 1995).

A general, typical LLD is described in Table 4. Some available statistics were used, on the consumption of bread, pulses, potatoes, milk, meat and fish between 1930 and 1950 (CBS 2001). Due to the low income situation, no food was wasted (Jobse-Van Putten 1995). This LLD is plant based and includes a maximum energy percentage (en%) of 30% deriving from an animal origin. In the countryside most of the people were self-sufficient. They had a vegetable garden and an acre with rye or other grains and a pig. Once a year the pig was slaughtered and the pork and grease was consumed until lent. Bread was baked once a week. Farmers in the lower grassland -clay and moor- areas (river sides, lake sides and coastal area) owned cows. One cow was available per four inhabitants. The butter was mostly sold and buttermilk was for own use. Porridge cars went by homes on a daily basis to sell porridge made of buttermilk and barley. People living in the eastern sand areas consumed a more rye based diet. In the cities more wheat was consumed. Typical for the culture is the cooking of hot meals in one pot (‘*stamppot’*) and for several days. Thanks to the trade connections, rice, spices, coffee and tea were common at an early stage. Fresh fish was only available in coastal zones; fresh water fish consumption was dropped in the 18th century, due to overfishing. Beef, poultry and game were not regular on the menu.

The Health Score of 106 and the Sustainability Score of 97 of the LLD (Table 3) are comparable to the recommended guidelines (Health Council 2006), but the Health Score is lower and the Sustainability Score higher than the MD and NND. The scores of the traditional LLD are higher than the present diet. Now we look for an interactive method to improve the suboptimal intake of vegetables, fruits, fiber and fish and to lower the salt consumption in the LLD.

Table 3. GHGE, LU, Sustainability Score, nutritional characteristics, and Health Score of the present Dutch, Mediterranean, New Nordic, historical Low Lands, and optimized Low Lands diets (males 31-50y).

		dietary guidelines	**present Dutch**	**Mediterranean**	**New Nordic**	**historical Low Lands**	**optimized Low Lands**
1	GHGE (kg CO_2-eq)	3.07	3.52	3.24	3.82	3.24	**2.60***
2	Land use (m^2*year)	3.08	4.15	4.15	3.42	3.18	**2.86**
	Sustainability score	101	82	86	86	97	**115**
1	Vegetables (grams)	200	119	300	**420**	179	215
2	Fruits (grams)	200	82	250	**350**	145	277
3	Fatty acids (en%)	25	34.5	**20.5**	21.5	25.2	24.2
4	Saturated fatty acids (en%)	10	12.7	6.9	**6.7**	8.4	7.3
5	Trans-fats (en%)	1	1.0	0.5	0.4	0.3	**0.2**
6	Free/added sugars (en%)	10	10.0	7.6	9.4	**3.5**	4.5
7	Dietary fiber (grams)	40	23	33	**47**	37	40
8	Salt (grams)	6.0	7.9	**5.0**	5.0	7.6	5.8
9	Fish (grams)	37	12	37	**43**	25	37
10	Energy in diet (kcal)	2500	2647	2503	2560	2651	**2500**
	Health score	100	71	122	**137**	106	123

best scores in **bold*

4. Methods: Linear programming

Although it cannot be assumed that a healthy diet will always have lower GHGE (Macdiarmid 2013), we do expect that linear programming will make it possible to find a diet with lower impacts than those diets found in literature. This is confirmed in a paper that is currently under submission (Van Dooren et al. 2014b). Several studies in other countries - for instance UK, France and New Zealand - have also successfully used linear programming for diet optimization.

Linear programming is a mathematical technique that allows the generation of optimal solutions (Dantzig G and M. 1997). The method we use for linear programming in this study is very similar to the one used by (Macdiarmid et al. 2012). This mathematical method optimizes an outcome which is a linear function of several variables that can be controlled (e.g. the amount of food eaten), while subject to a number of constraints (e.g. dietary requirements) (Macdiarmid et al. 2011). The linear programming algorithm minimizes the absolute changes in terms of portions to the present diet (see also Maillot et al. 2010); weighted by a proxy of popularity, while satisfying a number of constraints. Due to the weighing, it penalizes positive deviations from the present diet differently from negative deviations. More specifically the optimization weights are constructed from the normalized value of the total food consumption based on weight (n=206, based on (Van Rossum CTM et al. 2011)) for male adults (31-50y). In other words: Diets are improved by keeping as much as possible products from the starting point, and by adding products that are popular at the moment, in order to create a culturally acceptable diet.

The total GHGE of a diet consisting of amounts of n food products (x_1, x_2, ., x_n) and the associated GHGE of each food product per unit weight ($GHGE_i$) is as follows: Eq.3.

$$GHGE_{diet} = \sum_{i=1}^{n} x_i GHGE_i \qquad \textbf{Eq. 3}$$

In addition, the diet has to satisfy the energy and nutrient requirements (constraints) and, when applicable, an upper limit for total GHGE. Each constraint can be denoted as b_1, b_2, .., b_m, and with each food product i contributing a_{ij} per unit weight to requirement j, a set of j dietary constraints was established such that: Eq. 4.

$$b_j \geq \sum_{i=1}^{m} a_{ij} x_i \qquad \textbf{Eq. 4}$$

Optimization is done by linear programming using the newly developed Optimeal® software of Blonk Consultants and the Netherlands Nutrition Centre, which runs with MATLAB Compiler 7.16 and Microsoft Access Runtime. During optimization we use 33 nutrients and GHGE as constraints. Adequate intake levels and dietary guidelines of 33 nutrients are given by the Netherlands Nutrition Centre (2008) and the Health Council of the Netherlands (Health Council 2001; 2006; 2009) as the first constraint. The 33 nutrients and nutritional indicators are: energy, protein, carbohydrates, total fat, saturated fatty acids, trans fatty acids, polyunsaturated fatty acids, cholesterol, dietary fiber, alcohol, water, sodium, calcium, phosphorus, magnesium, iron, copper, selenium, zinc, iodine, retinol, vitamin D, vitamins B1, B2, B6, B12, folic acid, vitamin C, n-3 fatty acids (EPA + DHA), fruit consumption and vegetable consumption. Secondly, the upper boundary for climate impact (GHGE) is set to 2.60 kg CO_2eq/day. This represents a 20% reduction of the present GHGE of the Dutch diet (males 3.52 kg CO_2eq/day (Marinussen et al. 2012)), as feasible target.

5. Results: an optimized LLD

Optimization of the historical LLD resulted in an increased content of vegetables, fruits and dietary fibers, and a decrease in salt. Table 4 shows the result of the historical research into the LLD and the results of optimization by linear programming. Due to the constraints and penalty rules, the optimization resulted in a diet which is palatable and in line with the traditional diet. Various elements of this diet are in use at the moment, for instance a high consumption of potatoes and wheat bread compared to other European countries, the preference for cabbage, root vegetables and local fruits (apple, pear), the level of milk consumption and the habits of drinking coffee, tea and beer.

Table 4: Quantification of the historical and optimized Low Lands Diet for male adults, by historical research and linear programming.

historical Low Lands diet	**optimized Low Lands diet**
350 g potatoes	350 g potatoes
180 g vegetables (leafy, roots, cabbages)	215 g vegetables (extra lettuce and kale)
145 g fruits (apple, pear, berries)	277 g fruits (extra pear)
18 g legumes	51 g legumes
	24 g nuts
35 ml soup	35 ml soup
210 g rye and wheat bread (6 slices)	210 g rye and wheat bread (6 slices)
30 g rice and 10 g other grain products	10 g other grain products
2 portions porridge incl. 250 ml buttermilk and 150 ml full fat milk	2 portions porridge incl. 325 ml skimmed milk and 150 ml full fat milk
20 g cheese	no cheese
40 g butter, rapeseed oil, margarine	40 g butter, rapeseed oil, margarine
25 g fish	37 g oily fish
55 g meat (pork, beef and chicken)	48 g meat (pork and chicken)
300 ml coffee, 250 ml tea	300 ml coffee, 250 ml tea
300 ml beer, 6 ml wine	300 ml beer
	300 ml water
32 g extras (sugar, jam, chocolate, syrup, cake)	32 g extras (sugar, jam, chocolate, syrup, cake)

Furthermore, this optimization placed the LLD scores in between the Health Scores of the MD and NND, but resulted in the highest Sustainability Score (i.e. lowest environmental impact, Table 3). GHGE is 2.60 kg CO_2-eq per day and LU 2.86 m^2*year. The sustainability scores (Table 3) of the Nordic, Mediterranean, traditional LLD and recommended dietary guideline are on the same level of about 100. Only the optimized LLD is higher. An unexpected result is that the Health Score of the optimized LLD is close to the high score of the NND. Both are higher than the MD, which is generally considered as very healthy.

The optimization results in a diet within the 33 nutritional constraints and with low GHGE and LU (or high Sustainability Score). To reach this diet, the consumption of (local and seasonal) fruits, vegetables and legumes has to increase. Nuts should be added to the diet. The consumption of fish and white meat should be increased and the intake of cheese and beef has to be reduced. The optimized LLD seems to be a cultural acceptable option, which amply meets the nutritional and environmental constraints.

6. Discussion

In this paper we were able to define a nutritionally healthy, environmentally friendly, and culturally acceptable Low Lands Diet. The historical quantification of the diet before 1940 is subject to uncertainties. Changes in starting point will result in other optimal solutions. The qualities of the LLD are in line with the benefits of MD and LLD, based on the quality of historical diets from decades ago. In order to give relevant recommendations both historical, cultural patterns and present consumption have to be taken into account.

Both gastronomists and nutritionists are beginning to believe that there is a shared route to creating regional diets and an opportunity to develop a healthy diet that bridges gastronomy, health and sustainability (Mithril et al. 2012). Eating according to the Dutch dietary guidelines is one way to get to a healthier and a lower environmental impact than the present Dutch diet (Health Council 2011). However, further improvements in health scores, GHGE, and LU are within reach, we concluded in (Van Dooren et al. 2014a). We demonstrated that the MD is generally the health focus option with a high Sustainability Score. On the other hand, this paper demonstrates that the NND is also a prototype regional diet taking health, food culture, palatability and the environment into account. The principles and guidelines can indeed be applied in any region (Mithril et al. 2012). We demonstrated the possibility to apply this to the Low Lands.

This study confirms that if we combine the strong points of historical diets, with optimization on health and sustainability parameters, then even a local diet with higher scores is possible. Although the NND has the highest Health Score, a higher Sustainability Score is found in the optimized LLD.

Earlier, Swedish (Livsmedelsverket 2009) and Finnish governments (Steering Group 2010) have put together committees to give policy advice on environmentally effective diets. In line with these recommendations, four simple considerations for environmental impact were used in the formulation of the NND:

- Focus on locally grown foods.
- Focus on foods from organic food production.
- Focus on composing a proportion of the diet from foods sourced from the wild countryside, encouraging biodiversity and minimizing use of fertilizers and pesticides.
- Focus on minimizing waste and utilizing all of every food purchased (Mithril et al. 2012).

Saxe et al. (2013) confirmed that the climate impact of the local, organic NND is lower than the average and recommended diets. The four considerations mentioned above are also transferrable to and in use in the Low Lands, although the size of wild countryside in the Low Lands is much smaller, even a century ago. These Scandinavian recommendations and guidelines have a qualitative character. In this study we have quantified the sustainability in a combined score of GHG and LU. We also demonstrate that the NND is not optimal in the Sustainability Score. The reason is a relative high amount of meat, fish, vegetables and fruits. The optimized LLD showed that with smaller quantities of these products comparable nutritional health can be achieved. It is important to mention the common ground of the results for MD, NND and LLD. All diets consist of basic food products, which are high in nutrient density and most of the time low in energy density. Most of the volume is plant based products, such as whole grains, fruits, vegetables, oils, pulses and nuts. All include animal products in moderate quantities.

When interpreting the results it is important to consider that the methodology used to score health and sustainability is based on a limited number of parameters. Other parameters, such as fossil energy use, water use, and ReCiPe-score were outside the scope of this study. The data are not based on dietary assessments, but on a general description from the literature about different diets. This results in some uncertainties. Sustainability scores are based on Dutch data on LCAs of common food products. Most of these products are familiar to the Mediterranean and Nordic cuisine, but difference in transport and agricultural methods would result in slightly different calculations of GHGE and LU. On the other hand, the results reflect a hypothetical situation where these NND and MD are transferred to the Low Lands. Because of the limited number of products (206) used in the calculations, no detailed results of the diets could be given. For instance some traditional products were missing and replaced by comparable products (e.g. oats and barley by wheat). Furthermore, the study should not be interpreted as a retrospective study on the sustainability impact of the situation before 1940. Surely, energy intensity, processing and yields differ significantly over time. The calculation of the traditional LLD reflects the situation if male adults from our area were to eat as our ancestors and grandparents did 80 years ago. Rather than trying out diets recommended in the literature, this study confirms the usefulness of linear programming to improve existing or culturally relevant diets.

7. Conclusion

An optimized Low Lands Diet has the same nutritional characteristics as the Mediterranean and the Nordic Diet, and results in a lower environmental impact and a higher sustainability score. These results are relevant because an adaptation of the historical diet, which fits better into the present eating habits, climate and agricultural tradition of the Low Lands, is assumed to be easier to achieve than a transition into a foreign Southern or Northern European diet.

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Optimising land use and consumption of livestock products in the human diet with an increasing human population in the Netherlands

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ABSTRACT

Land use related to food production is generally quantified using product-based life cycle assessments. We, however, quantified land use of diet scenarios with a land use optimization model. Energy and protein requirement of human populations, varying from 15 to 30 million people, were met with the consumption of arable crops, meat and milk. Diet scenarios explored contained 0% up to 80% of animal protein. First findings show that a limited contribution of animal protein in the diet resulted in lowest land use per capita, because of utilization by livestock of products inedible for humans. Moreover, utilization of land unsuitable for crop production provides the opportunity to sustain a larger population compared to a situation in which these soils were not used to produce food.

Keywords: human diet, land use, land optimization modelling, crop production, animal production

1. Introduction

Currently, about 40% of all terrestrial land area in the world is used for agriculture (FAOSTAT 2013). Non-agricultural land area consists of (tropical) forest, desert, mountains and tundra, but also of cities, industrial areas and roads. Pressure on land will increase, because of increasing land degradation (Stringer 2008) and increasing demands for food, biofuel, biomaterials, housing and infrastructure for an increasing population of 9.6 billion in 2050 (United Nations 2013). Simultaneously, there is need for preservation of nature (Smith et al. 2010, The Royal Society of London 2009, TheWorldBank 2007).

One way to feed an increasing world population with a limited land area is to increase productivity per hectare, commonly referred to as sustainable intensification (Tilman et al. 2011). Other ways to feed an increasing world population with a limited amount of land area include reducing the amount of food waste along the chain (Gustavsson et al. 2011, Kummu et al. 2012) or a change in human diets towards food products requiring less land area (Meier & Christen 2013).

Various scientific studies have assessed the impact of a change in human diet on land use (Collins & Fairchild 2007, Gerbens-Leenes & Nonhebel 2002, Peters et al. 2007, Rabbinge & Van Latesteijn 1992, Risku-Norja et al. 2008, Stehfest et al. 2009, Thibert & Badami 2011, Wirsenius et al. 2010). These studies support the general conclusion that plant-based diets require less land than animal-based diets (Meier & Christen 2013, Stehfest et al. 2009, Van Kernebeek et al. 2014, Wirsenius et al. 2010). Some studies even propagate to substitute ruminant meat by monogastric meat to lower land area requirements for food production (Collins & Fairchild 2007, Stehfest et al. 2009, Wirsenius et al. 2010). However, compared with diets of ruminants, diets of pigs and poultry are relatively rich in products such as cereals, that humans could consume directly (De Vries & De Boer 2010). Moreover, in a situation where arable land is limiting, ruminant production on land being unsuitable for crop production may contribute to global food security (Peters et al. 2007).

To avoid competition between humans and animals for land and products, such as cereals, we could stimulate to feed by-products from crop cultivation or the food industry to animals, and to feed grass from land less suitable for crop cultivation to ruminants. Peat soils in the Netherlands, for example, are not suitable for production of arable crops, such as grain or potatoes, but are valuable for grass-based livestock production. The objective of this paper is to examine the relative contribution of livestock production in a human diet, with an increasing human population, given a limited land area in a region. We used the Dutch agricultural system with different population scenarios as an illustration, assuming no import and export of food and feed.

2. Methods

To fulfil our research objective, we assessed agricultural land area required to feed a growing population in the Netherlands with various dietary choices, using linear programming. We varied population sizes from 15 million people, which is the approximate current Dutch population size, up to the maximum number of people that could potentially be fed, with steps of 5 million people. Diet scenarios ranged between 0% animal protein in the diet up to 80% animal protein, with steps of 5%.

The system studied was directed at production of food (i.e. energy and protein) for a given human population in the Netherlands, assuming no import and export of food and feed. We distinguished cultivation of food crops (grains, root- and tuber crops, oil crops and legumes), feed crops (maize silage) and grass. Crops could be cultivated on clay and sandy soils. Peat soils could be used for cultivation of grass. Our system also included dairy and pig production. Land area for animal production consisted of feed production. In the modelling study, we assumed typical, current agricultural yields for both crop and animal production (PPO 2009, 2012, Vermeij 2012). We excluded products from fisheries and aquaculture as these are not land-based systems. We also excluded products from greenhouses as these do not require agricultural land. For the same reason, we did not account for land required for infrastructure and production sites for, e.g., artificial fertilizer and machinery.

The objective function of our optimization model was to minimize the use of land for the defined crops and grassland on clay, sand and peat soils, whilst meeting human nutritional requirements. Current existing agricultural land area in the Netherlands was set as physical boundary.

Daily per capita human requirements were defined as 2000 kcal, 57 grams of protein and a maximum of 90 grams of total sugars (EFSA 2009, 2012).

2.1. Production of crops and crop products

Crop production and yields were based on typical current agricultural practices in the Netherlands. We distinguished between crop production on clay, sand and peat soil. We used multiple-year average yields for each soil type. Apart from differences in yields across soil types, no further geographical differences in yields and inputs were assumed, as the Netherlands is a small country with a relatively homogeneous climate. We accounted for 2-7 percent land area requirement for the production of seeds and tubers.

We distinguished four groups of food crops: grains, root- and tuber crops, oil crops and legumes. From each group, we included the crop with the largest land area in the Netherlands as a model crop, except for the group of root- and tuber crops, where we included two model crops. We included wheat as model crop for grains, potatoes and sugar beet for root- and tuber crops, rapeseed for oil crops and brown beans (Phaseolus vulgaris) for legumes. In addition, we included maize silage and grass as forage for dairy cattle.

Crops are grown in crop rotations. We used rotation schemes as provided by Van Ittersum et al.,(1995). Grass and silage maize were considered as mono crop rotations, whereas the other crops were considered in four to six year rotations.

Agricultural area in the Netherlands totals 1,842 x 10^3 hectares in 2012 (CBS 2013). Based on the land use map by Lesschen et al. (2012), we attributed the total agricultural area to three soil types:

779 10^3 hectares of clayey soils, 839 10^3 hectares of sandy soils, and 224 10^3 hectares of peat. Due to the vulnerability of peat to soil subsistence and oxidation after tillage, we assumed that these soils can only be used for grass production to feed cows.

We included various post-harvest processes to convert harvested products into human edible products and feed ingredients. Processes included were: dry milling of wheat, wet milling of potato, peeling of potato, sugar processing and crushing of oil seeds. The nutritional value of the output products were taken from (PDV 2011) for feed and (RIVM 2013) for food products.

2.2. Animal production

We modelled the pig and dairy production system using the concept of an animal production unit (PU), based on typical current management (Vermeij 2012). A PU consisted of one animal unit per year for an average Dutch farm, including associated breeding animals and offspring. Per PU, we defined output in terms of meat and milk. We computed associated nutritional requirements in terms of energy, protein and fiber. Moreover, other feeding constraints, e.g. daily intake capacity, were defined. The optimization model, finally, composed the feed ration.

For the pig production system, a PU comprised a fattening pig unit, including the associated rearing gilts and breeding sows. Per PU per year, 3.26 fattening pigs were slaughtered, producing 170 kg pork per year.

For the dairy production system, a PU comprised one milking cow, including associated young stock. A Dutch dairy cow PU produced on average 8120 kg milk (8586 kg FPCM; Fat and Protein Corrected Milk) and 141 kg meat per year (LEI 2013, PDV 2012) available for human consumption.
We assume no impact of ration composition on production parameters such as protein content in milk, dressing percentage and carcass quality.

2.3. Losses and waste

Losses and waste of arable and animal products were modelled on weight basis using Gustavsson et al. (2011) and Biewenga et al. (2009). These include losses in the feed chain during conservation and feeding, losses during transport and distribution, and waste by the consumer.

3. Results

The first findings of our analysis show the agricultural land area for diets varying in percentage of animal protein for various population sizes (Figure 1).

Diets containing no animal protein had slightly higher land use compared to diets that contained a modest amount (5-30%) of animal protein. This is due to the utilization by animals of products that cannot be consumed by humans, such as wheat straw and sugar beet molasses. At higher levels of animal protein, additional feed crops are produced, which results in increasing land use. Besides, with increasing percentage of animal protein, peat soils are taken into use for the production of grass for dairy cows.

The figure also shows that, at higher population numbers, a vegan diet cannot provide sufficient food to sustain the population. Crop production on clay and sandy soils does not produce sufficient protein for this large population size. Due to the utilization of peat soils by dairy cows, an extra number of people can be fed potentially.

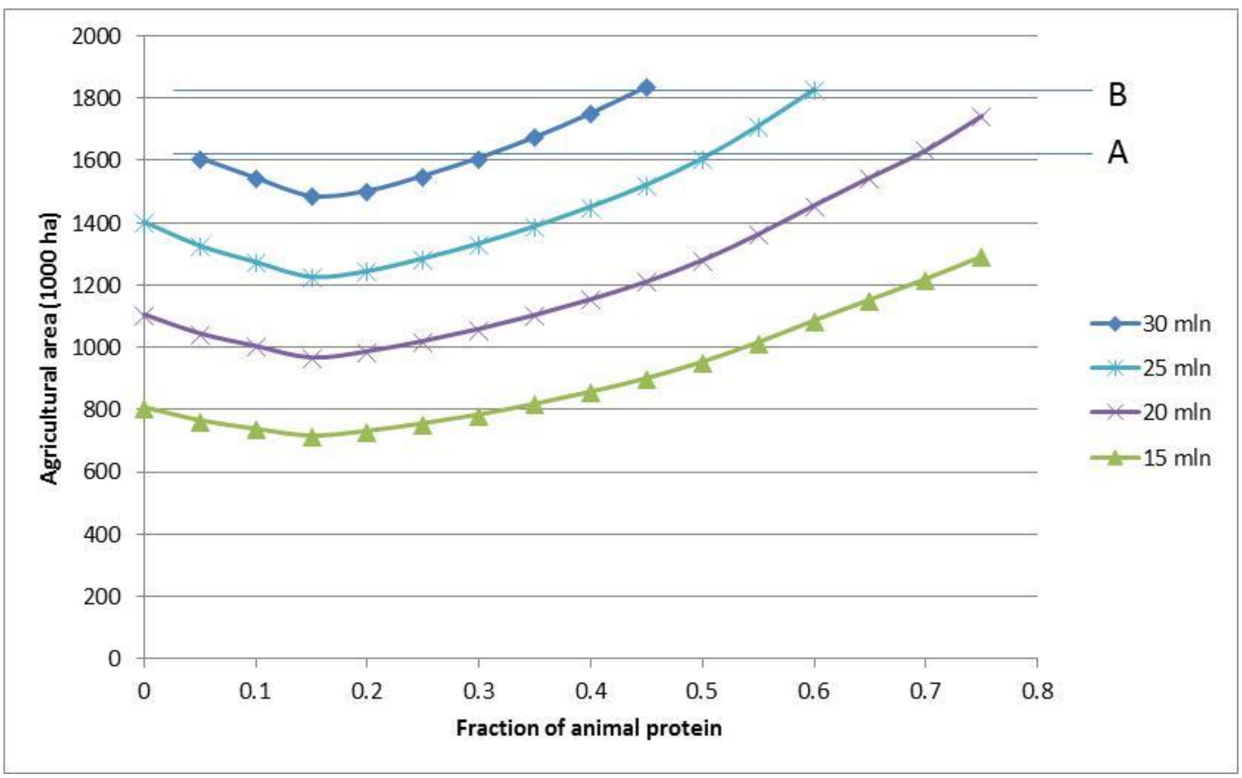

Figure 1. Agricultural land use (ha) in the Netherlands with increasing population, for diets differing in the amount of animal protein. The current Dutch population is 17 million. The Netherlands has around 1.8 x 10^6 hectares of agricultural land; we assumed no import of food. A: total agricultural area on clay and sandy soils. B: total agricultural area on clay, sand and peat soils.

4. Discussion

Our analysis was limited to land use. We did not consider economic effects of dietary choices. Besides, land area requirement is only one of many impact categories that should be accounted for when assessing overall sustainability of human diets. In the Netherlands, high crop yields partly result from high levels of inputs, which may reduce required land area at the cost of long-term sustainability in relation to, e.g., ecosystem services (Foley et al. 2005). Western diets contain in the order of 70% animal protein (Van Kernebeek et al. 2014). A reduction of animal consumption in affluent diets could help reducing land use, as well as greenhouse gas emissions (Hedenus et al. 2014).

Because we do not use allocation in our optimization model, it was not possible to express land use per kg of product or per kg of protein, as is often seen in literature (De Vries & De Boer 2010). However, we were able to compare land use of the diets with results from literature (Meier & Christen 2013, Terluin et al. 2013). Land use required for diets reported by Terluin et al. (2013) varied between 470 and 1000 m^2/capita/year. Land use required for diets reported by Meier and Christen (2013) varied between 1000 and more than 2000 m^2/capita/year. Land use required for diets in our study varied between about 500 and 1000 m^2/capita/year.

5. Conclusion

The ability of animals to utilize products that cannot be consumed by humans can help to reduce land use for food production. Moreover, utilization of land unsuitable for crop production, in our case peat soils, provides the opportunity to sustain a larger population compared to a situation in which these soils are not used to produce food.

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Integrated modeling of feeding and breeding strategies to reduce greenhouse gas emissions along the production chain of milk

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ABSTRACT

We evaluated the impact of feeding and breeding strategies to reduce greenhouse gas (GHG) emissions from dairy farming. A whole-farm optimization model was combined with mechanistic modeling of enteric fermentation and LCA to determine the impact on GHG emissions and farm income. Feeding strategies included supplementation of extruded linseed (56% linseed; 1 kg/cow/day in summer, 2 kg/cow/day in winter), supplementation of nitrate (75% nitrate; 1% of dry matter intake), and reducing grass maturity. In case of breeding, the impact of one genetic standard deviation improvement in milk yield (687 kg/cow/year) and longevity (271 days) was assessed. Supplementing linseed reduced emissions by 9 kg CO_2-equivalents (CO_2e)/ton fat-and-protein-corrected milk (FPCM), supplementing nitrate by 32 kg CO_2e/ton FPCM, and reducing grass maturity by 11 kg CO_2e/ton FPCM. All strategies reduced farm income. Increasing milk yield and longevity reduced emissions by 27 and 23 kg CO_2e/ton FPCM, respectively. Both strategies increased farm income.

Keywords: dairy farming, greenhouse gas mitigation, linear programming, economic analysis, life cycle assessment

1. Introduction

Dairy cattle, producing milk, meat, and non-edible products (e.g. manure), are responsible for about 30% of global greenhouse gas (GHG) emissions produced by the livestock sector (Gerber et al., 2013). About half of these GHG emissions is related to enteric fermentation. Other important sources of GHG emissions are feed production and manure management. Two important areas of interest to reduce GHG emissions from dairy farming are feeding strategies to reduce emissions from enteric fermentation, and breeding strategies to improve animal productivity.

Examples of feeding strategies to reduce emissions from enteric fermentation are dietary supplementation of fatty acids, dietary supplementation of nitrate, and reducing the maturity stage of grass herbage and grass silage (Martin et al., 2008; Van Zijderveld et al., 2011; Brask et al., 2013). Examples of breeding strategies to improve animal productivity are increasing milk yield and longevity (Bell et al., 2010). Increasing milk yield per cow reduces GHG emissions per kg milk by diluting emissions from production and fermentation of feed related to maintenance. Increasing longevity reduces GHG emissions per kg milk by reducing the number of female replacements contributing to GHG emissions during maintenance and growth, without producing milk, and by increasing lifetime milk yield per cow, diluting emissions related to rearing.

Most studies that evaluate the potential of feeding and breeding strategies to reduce GHG emissions from dairy farming do not account for emissions other than enteric methane (CH_4), do not account for changes in farm management to adapt the farm optimally to the particular strategy, or do not account for consequential effects in other parts of the milk-production chain. To understand which strategies can contribute to reducing the net contribution of dairy farming to global GHG emissions, an integrated approach is required that accounts for all changes in farm management and includes all changes in GHG emissions along the chain.

The aim of this study was to analyze the impact of several feeding and breeding strategies to reduce GHG emissions from dairy farming on GHG emissions at chain level (i.e. from cradle to farm-gate) and on labor income at farm level using an integrated approach. We combined a whole-farm optimization model with a mechanistic model for enteric methane and life cycle assessment (LCA). Feeding strategies under study included dietary supplementation of extruded linseed, dietary supplementation of nitrate, and reducing the maturity stage of grass herbage and grass silage (for further details, see Van Middelaar et al., 2014a). In case of breeding, we focused on the impact of one genetic standard deviation improvement in milk yield and longevity (for further details, see Van Middelaar et al., 2014b). By evaluating the impact of one unit change in individual traits, the relative value of each trait to reduce GHG emissions along the chain can be determined.

2. Methods

2.1. Dairy farm linear programming model

A dairy farm linear programming (LP) model based on Berentsen and Giesen (1995) was used to simulate a Dutch dairy farm before and after implementing the strategies. The farm production plan was optimized based on the objective to maximize labor income of the farm family. The LP model is a static year model and includes all relevant activities and constraints that are common to Dutch dairy farms, such as on-farm roughage production, purchase of feed, and animal production, including the rearing of young stock. Prices of purchased and sold products were based on KWIN-V (2008).

The central element of the LP model is an average dairy cow from the Holstein Friesian breed, with a given milk production and longevity, calving in February, and conditions representing the dairy cattle of the farm. Feed requirements (energy and protein) and intake capacity of this average cow were determined using the bio-economic model of Groen (1988). The same model was used to determine herd composition and yearly replacement rate, based on the average longevity of the cow. The replacement rate determines the number of young stock that needs to be kept on the farm for yearly replacement of the dairy cows.

The model distinguishes a summer and a winter period regarding feeding. Available land can be used as grassland or as corn land. Dietary options include grass from grazing, grass silage, corn silage, and three types of concentrates that differ in protein levels (i.e. standard, medium and high). All dietary options were available in winter and summer, except for fresh grass (only in summer). Table 1 shows the feed characteristics of the feeds that are standard available in the model. Based on feed restrictions, the LP model matches feed requirements of the cow with on-farm feed production and purchased feed. Constraints of the model include fixed resources of the farm (e.g. land area, family labor), links between activities (e.g. fertilizer requirements of grass- and arable land with available nutrients from manure and purchased fertilizers), and environmental policies (e.g. limits to the application of nitrogen (N) and phosphate (P_2O_5) fertilization). For a more detailed description of the LP model see Van Middelaar et al. (2013).

Table 1. Feed characteristics of feeds standard available in the dairy farm LP model, expressed per kg dry matter (DM).

Feed product	NE_L [1] (MJ/kg DM)	DVE [2] (g/kg DM)	OEB [3] (g/kg DM)	N [4] (g/kg DM)	Fill value [5] (kg/kg DM)	NDF [6] (g/kg DM)	Crude fat (g/kg DM)
Concentrates							
- standard protein	7.21	100	6	24.1	0.29-0.72	414	48
- medium protein	7.21	133	28	32.2	0.29-0.72	407	51
- high protein	7.21	200	83	48.3	0.29-0.72	312	46
Dietary urea	0.00	0	2920	467.0	0.00	0	0
Fresh grass normal cut (1700 kg DM/ha)							
- 125 kg N	6.62	94	9	28.0	0.93	457	37
- 175 kg N	6.68	96	16	29.4	0.93	452	38
- 225 kg N	6.73	98	23	30.9	0.93	448	39
- 275 kg N	6.77	99	31	32.4	0.93	445	40
Grass silage normal cut (3500 kg DM/ha)							
- 125 kg N	5.89	70	22	25.6	1.08	506	35
- 175 kg N	5.93	71	31	27.4	1.08	501	36
- 225 kg N	5.97	73	39	29.0	1.08	497	37
- 275 kg N	6.00	74	47	30.6	1.08	493	39
Corn silage	6.56	58	-36	13.4	1.02	373	25

[1] Net energy for lactation. [2] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994). [3] Rumen degradable protein balance according to Dutch standards (Tamminga et al., 1994). [4] Nitrogen. [5] Fill value per kg DM feed expressed in kg of a standard reference feed (Jarrige, 1988). The fill value of concentrates increases with an increase in concentrate intake. [6] Neutral detergent fiber.

2.2. Calculating greenhouse gas emissions

We used LCA to calculate emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) from the different stages along the production chain, up to the moment that milk leaves the farm gate. Processes included are the extraction of raw materials to produce farm inputs, the manufacturing and distribution of these inputs, and all processes on the dairy farm. Stages related to transport and processing of milk were assumed to be unaffected by the strategies, and, therefore, not included in the analysis.

Emissions from the production of synthetic fertilizer, pesticides, tap water, and energy sources (gas, diesel, and electricity) were based on Eco-invent (2007), and from the production of concentrates and milk replacer on Vellinga et al. (2013). Emissions from production of concentrates include emissions from the production of inputs (e.g. fertilizers, pesticides, machinery, and energy), direct and indirect N_2O emissions from cultivation, CO_2 emissions from liming and urea fertilization, emissions from drying and processing, and emissions from transport in between stages, up to the farm gate.

Emissions of CH_4 from on-farm processes relate to enteric fermentation and to manure management. Emission of enteric CH_4 from dairy cows was calculated using a mechanistic model originating from Dijkstra et al. (1992). Production of CH_4 is estimated from the hydrogen (H_2) balance. Sources of H_2 include H_2 from production of acetate and butyrate, and from microbial growth with amino acid as N-source. Sinks of H_2 include H_2 used for production of propionate and valerate, for microbial growth with ammonia as N-source, and for biohydrogenation of lipids. The surplus H_2 is assumed to be completely converted into CH_4. To calculate the effect of dietary supplementation of extruded linseed and nitrate on enteric CH_4 production, additional calculations were required (see section 2.3). In case of breeding, results of the mechanistic model were transformed into empirical relations between dry matter (DM) intake of feed ingredients and CH_4 emission factors per ingredient (Vellinga et al., 2013). For young stock, enteric CH_4 emission was based on IPCC Tier 2 methods and default values, i.e. the average gross energy content of feed is assumed to be 18.45 MJ/kg DM, and 6.5% of the gross energy intake is converted to CH_4 (IPCC, 2006). Emissions of CH_4 from manure management were based on national inventory reports, i.e. 0.746 kg CH_4 per ton manure produced in stables, and 0.110 kg CH_4 per ton manure produced during grazing (De Mol and Hilhorst, 2003).

Emissions of CO_2 from on-farm processes related to the combustion of diesel and gas were based on Eco-invent (2007). Emissions of N_2O from on-farm processes include both direct and indirect N_2O from manure management and from N application to the field, including N from manure, synthetic fertilizers, and crop residues. Indirect N_2O emissions result from N that is removed from the farm via leaching of nitrate and volatilization of ammonia and nitrogen oxide (IPCC, 2006). Emissions of N_2O from crop residues were based on IPCC (2006). Other N_2O emissions were based on national inventory reports (see Van Middelaar et al., 2013).

Different GHGs were summed up based on their equivalence factor in terms of CO_2 equivalents (CO_2e) (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (IPCC, 2007). Emissions were calculated per ton fat-and-protein corrected milk (FPCM), i.e. milk corrected to a fat percentage of 4.0% and a protein content of 3.3%. After summing up emissions, they were allocated to the different outputs of the farm (i.e. milk and meat) based on the relative economic value of these outputs (i.e. economic allocation).

2.3. Feeding strategies

We evaluated the impact of feeding strategies for a current Dutch dairy farm on sandy soil. This farm has 44.9 ha of land, housing facility for 76 dairy cows with young stock, and a milk quota of 603 tons per year. Milk production per cow was assumed to be constant at 7968 kg/year (4.39% fat and 3.52% protein). Data were based on the Farm Accountancy Data Network of the Agricultural Economics Research Institute from the Netherlands (FADN, 2012). The maximum fresh grass intake in summer was assumed to be 12 kg DM/cow per day, because limited grazing was applied. Safety margins for requirements of true protein digested in the small intestine and for rumen degradable protein balance were set at 100 g/cow per day. In addition, the option to feed dietary urea was included. The maximum amount of urea was limited by including a restriction on the amount of non-protein-nitrogen (NPN) equal to the amount of NPN in the diet supplemented with nitrate (see below). The reference situation (REF1), which includes no predefined feeding strategy, was determined by maximizing labor income for this current Dutch dairy farm. Subsequently, one of the three feeding strategies was introduced. Labor income of the farm was maximized again to determine diets and farm plan after implementing each strategy.

Feeding strategy LINS. Extruded linseed was added as a commercially available linseed product described by Dang Van et al. (2008), containing 56.0% crushed linseed, 21.0% wheat, 15.0% sunflower cake, 4.5% field beans, 2.0% butylated hydroxytoluene, 1.0% linseed oil, and 0.5% salt. Table 2 shows feed characteristics of this product. One kg product/cow per day was prescribed in the diet in summer and 2 kg/cow per day in winter (the product contains 0.9 kg DM/kg product). The effect of adding fatty acids in the form of extruded linseed on enteric CH_4 production was based on (Eq. 1) by Grainger and Beauchemin (2011):

$$y = - 0.102\ x \qquad \text{Eq. 1}$$

where, y is the reduction in enteric CH_4 (g/kg DM intake); and x is the amount of dietary fat added (g/kg DM).

Feeding strategy NITR. A nitrate source ($5Ca(NO_3)2{\cdot}NH_4NO_3{\cdot}10H_2O$; 75 % NO_3 in DM) was added at 1% of dietary DM. Table 2 shows feed characteristics of this nitrate source. The effect of nitrate on enteric CH_4 production was based on Van Zijderveld (2011). Stoichiometrically, a reduction in CH_4 of 0.258 g/g nitrate is expected. In vivo, efficiency of CH_4 reduction decreases with increased levels of nitrate intake according to Eq. 2.

$$y = - 0.17\ x + 1.13 \qquad \text{Eq. 2}$$

where, y is the actual reduction in enteric CH_4 expressed as a fraction of the reduction potential according to stoichiometry; and x is the amount of nitrate expressed in g/kg metabolic weight ($kg^{0.75}$) per day. The metabolic body weight of the cow is assumed to be 129 kg.

Feeding strategy GMS. Reducing the maturity stage of grass herbage and grass silage results in a lower DM yield/ha per year, but increases grass quality in terms of energy and protein content per kg DM. Total yield in MJ net energy for lactation (NE_L)/ha per year was assumed to remain unchanged. In the reference situation, grazing was applied at 1700 kg DM/ha, and harvesting at 3500 kg DM/ha. After implementing the strategy, grazing was applied at 1400 kg DM/ha, and harvesting at 3000 kg DM/ha. Table 2 shows feed characteristics of less mature grass and grass silage (based on CVB, 2011). Costs per grass cut were assumed to be the same as in the reference situation. Due to a lower DM yield per grass cut, the number of cuts per year increased.

Table 2. Feed characteristics of feeds available after implementing the feeding strategies, expressed per kg dry matter (DM).

Feed product	**NE_L [1] (MJ/kg DM)**	**DVE [2] (g/kg DM)**	**OEB [3] (g/kg DM)**	**N [4] (g/kg DM)**	**Fill value [5] (kg/kg DM)**	**NDF [6] (g/kg DM)**	**Crude fat (g/kg DM)**
LINS							
Extruded linseed product	10.51	96	87	36.9	0.29	209	236
NITR							
Nitrate	0.00	0	1170	187.3	0.00	0	0
GMS [5]							
Fresh grass early cut (1400 kg DM/ha)							
- 125 kg N	6.67	96	10	28.9	0.93	442	37
- 175 kg N	6.72	98	18	30.4	0.93	437	39
- 225 kg N	6.77	100	26	31.9	0.93	433	40
- 275 kg N	6.82	102	35	33.5	0.93	430	41
Grass silage early cut (3000 kg DM/ha)							
- 125 kg N	5.96	73	27	27.6	1.08	488	36
- 175 kg N	6.01	74	38	29.5	1.08	484	38
- 225 kg N	6.04	76	48	31.3	1.08	480	39
- 275 kg N	6.08	77	58	33.0	1.08	476	41

[1] Net energy for lactation. [2] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994). [3] Rumen degradable protein balance according to Dutch standards (Tamminga et al., 1994). [4] Nitrogen. [5] Fill value per kg DM feed expressed in kg of a standard reference feed (Jarrige, 1988). [6] Neutral detergent fiber.

2.4. Breeding strategies

The LP model was adapted to future production circumstances to allow exploration of economic and environmental consequences of selective breeding. We evaluated the impact of breeding strategies for a representative Dutch dairy farm on sandy soil for the year 2020, and with a cow that has the same characteristics as an average Holstein Friesian cow in 2013 (Table 3). The future farm has 85 ha, which is the estimated size of an average Dutch dairy farm in 2020 (Rabobank, 2009). In 2015, the milk quota system will be abolished in the EU, and, therefore, no milk quota was assumed. The number of cows on the farm is an outcome of the LP model, and restricted by a constraint that prescribes that all manure produced on the farm needs to be applied on the farm. Grass and corn yield per hectare were increased based on historical data analysis (Berentsen et al., 1996; Rijk et al., 2013). For the environmental policies, no changes in limits to the application of N are expected, whereas limits to the application of P_2O_5 are reduced according to the new standards for 2020 (Vierde Nederlands Actieprogramma Nitraatrichtlijn, 2009). Furthermore, prices of milk components and purchased feed products were adapted based on price prediction for 2020 (KWIN-V, 2013). Based on the assumption that farmers become more efficient in the future, safety margins for true protein digested in the small intestine and for rumen degradable protein balance were set to zero. The reference situation for evaluating breeding strategies (REF2) was determined by maximizing labor income for this future Dutch dairy farm.

To determine the impact of one genetic standard deviation improvement in milk yield or longevity, each trait was increased with one genetic standard deviation, while keeping the other traits constant. The genetic standard deviation for milk yield of the Holstein Friesian breed in the Netherlands is 687 kg/cow per year (standard deviation applies to milk yield of a mature cow), and for longevity it is 270 days (CRV, 2012). Using the model of Groen (1988), the effect of this change on average production, feed requirements, herd composition and replacement rate was determined. Increasing milk yield increased feed requirements (Table 3). Increasing longevity changed herd composition (i.e. more cows in later lactations), and decreased replacement rate and number of young stock. Due to an increase in the number of cows in later lactations, milk yield of the average cow increased and fat content of the milk decreased, while feed requirement of dairy cows increased (Table 3). The new data on milk yield, feed requirements, and replacement rate were incorporated in the LP model, and labor income of the farm was maximized again to determine diets and farm plan after implementing each strategy.

Table 3. Production traits and feed requirements per cow, and yearly replacement rate (repl. rate) of the dairy herd for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation.

	Production traits				**Feed requirements**			**Repl. rate**
	Milk yield kg/yr	Fat %	Protein %	Longevity # days	Energy GJ NE_L[1]/yr	Protein kg DVE[2]/yr	Intake capacity kg/yr	%
Reference	8758	4.32	3.51	2150	44,553	545	6009	27.0
Incr. milk yield	9445	4.32	3.51	2150	46,961	583	6137	27.0
Incr. longevity	8795	4.31	3.51	2420	44,712	547	6037	22.5

[1] NE_L: Net energy for lactation. [2] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994).

3. Results

3.1. Feeding strategies

Table 4 shows the diets of the dairy cows and farm plan for the reference situation of the current farm (REF1) and the situations after implementing the feeding strategies. In REF1, the maximum amount of fresh grass is fed in summer, because this is the cheapest way of feeding. Corn silage is added up to 6.59 kg DM/cow per day in combination with standard protein concentrates and dietary urea. As a result, minimum requirements for energy and rumen degradable protein are met within the limiting intake capacity. In winter, 2.86 kg DM grass silage/cow per day is fed, which is the amount of grass left for ensiling after grazing, in combination with 10.98 kg DM corn silage/cow per day. High protein concentrates and urea were added to meet requirements for energy, rumen degradable protein, and true protein digested in the small intestine. 70% of the farm land was used as grassland and 30% as corn land. Labor income of the farm family was €42,605 per year. GHG emissions added up to 840 kg CO_2e/t FPCM. The most important contributor was enteric CH_4 (52%), followed by emissions from manure (14%), on-farm feed production (13%), purchased feed products (10%), and fertilizers (8%).

Feeding strategy LINS increased the fat content of the summer diet from 35 g/kg DM (REF1) to 44 g/kg DM. As a result, total DM intake reduced. The amount of corn silage decreased and standard concentrates and urea were removed from the diet. In winter, dietary fat content increased from 32 g/kg DM in REF1 to 56 g/kg DM (LINS). As a result, the amount of corn silage decreased by almost 3 kg DM/cow per day, and urea was removed from the diet. The amount of high protein concentrates remained to fulfill requirements for true protein digested in the small intestine. Labor income reduced to €26,564 per year. This reduction is caused almost completely by the relatively high costs of the extruded linseed product compared to the costs of corn silage and concentrates. In total, GHG emissions decreased by 9 kg CO_2e/t FPCM. Emissions of enteric CH_4 from dairy cows decreased by 42 kg CO_2e/t FPCM. Due to a decrease in the amount of purchased corn silage, concentrates, and urea, emissions related to the production of these products decreased by 29 kg CO_2e/t FPCM in total. Emissions from the production of the extruded linseed product added up to 63 kg CO_2e/t FPCM.

Feeding strategy NITR resulted in a dietary NPN level of 37 g/cow per day in summer, and 31 g/cow per day in winter, being the maximum amount of dietary NPN allowed. As a result, urea was removed from the diet. No other dietary changes occurred. Due to an increase in dietary N content, the amount of N in manure increased.

Table 4. Diets and farm plan for the current dairy farm (REF1) and after implementing one of the three feeding strategies.

		REF1	**LINS**	**NITR**	**GMS**
Diet dairy cows – summer period (kg DM/cow per day)					
Grass herbage		12.00	12.00	12.00	12.00
Corn silage		6.59	6.07	6.59	6.62
Concentrates standard protein		0.88	-	0.88	0.78
high protein		-	0.04	-	-
Urea		0.02	-	-	0.01
Extr. linseed product		-	0.90	-	-
Nitrate		-	-	0.20	-
Diet is restricted by [1]		E,I,R	E,T	E,I	E,I,R
Diet dairy cows – winter period (kg DM/cow per day)					
Grass silage		2.86	2.86	2.86	2.75
Corn silage		10.98	8.14	10.98	11.09
Concentrates high protein		2.40	2.36	2.40	2.37
Urea		0.06	-	-	0.06
Extr. linseed product		-	1.80	-	-
Nitrate		-	-	0.16	-
Diet is restricted by [1]		E,R,T	E,T	E,T	E,R,T
Farm plan					
Dairy cows	n	76	76	76	76
Milk production	ton FPCM/farm/year	603	603	603	603
Young stock	unit [2]	25	25	25	25
Grassland 225 kg N/ha	ha	31.4	31.4	31.4	31.4
Corn land	ha	13.5	13.5	13.5	13.5
Synthetic fertilizer	kg N/ha	117	118	111	116
	kg P_2O_5/ha	8	7	7	10
Purchased corn silage	t DM	96	48	96	98
Purchased concentrates	t DM	55	43	55	53
Urea	t DM	1	-	-	1
Extr. linseed product	t DM	-	38	-	-
Nitrate	t DM	-	-	5	-
Labor income	€	42,605	26,564	37,142	42,142
GHG emissions	kg CO_2e/ton FPCM	840	831	808	829

[1] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity. [2] One unit includes 1 animal < 12 months and 0.96 animal > 12 months.

As a result, the amount of synthetic fertilizer decreased. No other changes in farm production plan occurred. Labor income reduced to €37,142 per year. This reduction is caused by the higher costs of dietary nitrate compared with urea. In total, GHG emission decreased by 32 kg CO_2e/t FPCM. Emission of enteric CH_4 from dairy cows decreased by 33 kg CO_2e/t FPCM. Producing nitrate instead of urea increased emissions by 3 kg CO_2e/t FPCM. Changes in other emissions were minor and relate to an increase in the N content of manure.

Feeding strategy GMS did not affect the amount of grass in kg DM/cow per day in the summer diet. Due to a higher energy content and a higher rumen degradable protein content per kg grass, however, the amount of concentrates and urea slightly decreased and that of corn silage slightly increased. Because total DM yield per ha grassland decreased, the amount of grass silage in the winter diet decreased. Corn silage slightly increased, while the amount of concentrates and urea remained unchanged. Due to a higher N and a lower P content in the diet, the amount of N in manure increased, while the amount of P decreased. This is reflected by a change in purchased fertilizers. Labor income reduced to €42,142 per year. This reduction is caused mainly by an increase in costs related to grassland management, resulting from an increase in the number of grass cuts per ha per year. In total, GHG emissions decreased by 11 kg CO_2e/t FPCM. GMS reduced emissions of enteric CH_4 from dairy cows by 10 kg CO_2e/t FPCM. Changes in other emissions were minor and relate to changes in the diet and an increase in the N content of manure.

3.2. Breeding strategies

Table 5 shows the diets of the dairy cows and farm plan for the reference situation of the future farm (REF2) and the situations after increasing milk yield and longevity. For the reference situation the following results apply. In summer the maximum amount of fresh grass is fed. Subsequently, corn silage in combination with a small amount of medium protein concentrates is added to meet requirements for energy and rumen degradable protein balance. In winter, the diet contains 2.7 kg DM grass silage per cow per day, based on the amount of grass remained after grazing. Again, corn silage in combination with medium protein concentrates is added. The reference situation has 168 dairy cows, 59.5 ha of grassland and 25.5 ha of corn land. The number of cows is based on the amount of manure that can be applied on the farm according to environmental legislation. In the reference situation, application standards on the amount of P_2O_5 were restricting. The area of grassland is exactly 70%, which is the minimum requirement for farms to comply with the derogation regulation that allows the application of 250 kg N/ha per year from animal manure, instead of 170 kg N/ha per year. Labor income is €115,050/year. GHG emissions added up to 796 kg CO_2e/t FPCM. The most important contributor was CH_4 from enteric fermentation (50%). Other important contributors were emissions from manure and from production of concentrates (both 13%).

Increasing milk yield by one genetic standard deviation resulted in an increase in the number of cows and an increase in the area of grassland at the expense of corn land. Changes in diets resulted from an increase in requirements for energy and protein, and from an increase in the area of grassland. Grassland increased because of the increase in the number of cows and P_2O_5 application standards being restricting (more P_2O_5 from animal manure can be applied on grassland than on corn land). In the reference situation, the costs of an increase in grassland at the expense of corn land were higher than the revenues of keeping more cows. After increasing milk yield, the revenues per cow increased, and outweighed the costs of an increase in the area of grassland at the expense of corn land. After increasing milk yield, the number of cows and grassland increased until application standards for N from animal manure became restricting. Total milk production at farm level increased to 1691 t FPCM/year, and labor income to €135,477. In total, GHG emissions decreased by 27 kg CO_2e/t FPCM. Increasing milk yield per cow reduced emissions per t FPCM by diluting emissions related to maintenance and young stock. In addition, emissions changed because of changes in diets and farm plan, e.g. emissions from the production of concentrates decreased, because the amount of concentrates in the diets decreased.

Increasing longevity by one genetic standard deviation reduced the replacement rate of the dairy herd (from 27.0% in REF2 to 22.5 % after increasing longevity). Similar to milk yield, this resulted in a situation where corn land was changed into grassland to increase to amount of P_2O_5 that can be applied on the field, and hence the number of dairy cows. Because of the reduced replacement rate, less young stock was kept, reducing manure production of the herd. As a result, the number of dairy cows increased to 182. Again, the application standard for N from animal manure limited a further increase of dairy cows. Total milk production at farm level increased to 1677 t FPCM/year, and labor income to €128,765. In total, GHG emissions decreased by 23 kg CO_2e/t FPCM.

Table 5. Diets and farm plan for the future dairy farm (REF2) and after implementing one of the two breeding strategies.

		REF2	Milk yield	Longevity
Diet dairy cows – summer period (kg DM/cow per day)				
Grass herbage		12.0	12.0	12.0
Corn silage		8.4	8.9	8.4
Concentrates medium protein		0.7	1.3	0.7
Diet is restricted by [1]		E,R	E,R	E,R
Diet dairy cows – winter period (kg DM/cow per day)				
Grass silage		2.7	5.0	4.2
Corn silage		8.0	8.9	8.4
Concentrates medium protein		6.5	4.6	5.0
Diet is restricted by [1]		E,R	E,R	E,R
Farm plan				
Dairy cows	n	168	171	182
Milk production	ton FPCM/farm/year	1543	1691	1677
Young stock	unit [2]	51	52	46
Grassland 225 kg N/ha	ha	59.5	67.9	67.4
Corn land	ha	25.5	17.1	17.6
Synthetic fertilizer	kg N/ha	107	113	112
	kg P_2O_5/ha	-	-	-
Purchased corn silage	t DM	207	396	381
Purchased concentrates	t DM	247	207	213
Manure application is restricted by [3]		P	aN, P	aN, P
Labor income	€	115,050	135,477	128,765
GHG emissions	kg CO_2e/ton FPCM	796	770	774

[1] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity. [2] One unit includes 1 animal < 12 months and 0.96 animal > 12 months. [3] The intensity of the farm is restricted by the possibility to apply manure. Manure application can be restricted by: tN = total mineral N; aN = N from animal manure; P = P_2O_5.

Due to a lower replacement rate, emissions related to young stock (mainly enteric CH_4) decreased by 12 kg CO_2e/t FPCM. Due to a change in the diets of dairy cows towards more roughage and less concentrates, emissions from production of grass and corn silage increased by 10 kg CO_2e/t FPCM (including on- and off farm production), whereas emissions from production of concentrates decreased by 20 kg CO_2e/t FPCM.

4. Discussion

Each feeding strategy reduced GHG emissions along the milk-production chain, but also reduced labor income. A negative impact on labor income reduces the likelihood of adoption by farmers, because profitability is often the main driver in decision making. While supplementing diets with nitrate resulted in the greatest reduction, a reduction in maturity of grass and grass silage resulted in the smallest reduction in labor income. Combining the impact on labor income with the impact on GHG emissions showed that this latter strategy is most cost-effective, and, therefore, offers most potential to be implemented on commercial farms.

Both breeding strategies reduced GHG emissions along the milk production chain while increasing labor income. The reduction in GHG emissions per ton FPCM were greater for milk yield than for longevity. The increase in labor income was also greater for milk yield. In this study, however, correlation between traits were not considered. Including correlations might change the balance between milk yield and longevity in favor of longevity, because production traits such as milk yield are negatively correlated with fertility and health traits, whereas longevity is positively correlated with these traits (Pritchard et al., 2013).

Breeding strategies affect GHG emissions in the long term. To evaluate the impact of an increase in milk yield and longevity, therefore, the model farm was adapted to future production circumstances without a milk quota. Differences in labor income between REF1 (current farm) and REF2 (future farm) are explained mainly by an increase in farm size, greater forage production per ha, and change in prices in case of REF2. Because only prices of important in- and outputs were changed, and because price predictions contain uncertainty, the impact of breeding strategies on labor income should be judged on their relative impact. Opposite to feeding strategies, breeding strategies resulted in an increase in income. Costs related to breeding strategies are covered by breeding organizations resulting in prices farmers have to pay for semen. There is no reason to assume that these prices will change depending on the strategy.

Emissions per ton FPCM in REF1 (current farm) are low compared with results in literature (De Vries and De Boer, 2010). This is mainly caused by the relatively high amounts of maize silage and low amounts of concentrates in the diets of dairy cows, partly because urea was used. In addition, unlike most other studies we used a model farm and calculated feed intake, which may differ from the actual intake and may increase the efficiency of the farm. Differences in emissions between REF1 and REF2 are explained by the higher productivity and efficiency in case of REF2, representing the technical and institutional setting of 2020 in combination with precision feeding (i.e. skipping safety margins for feeding protein). These results imply that future dairy farms can reduce their environmental impact in terms of GHG emission per ton FPCM when aiming for an increase in efficiency.

Results of the breeding strategies represent the impact of one unit change in milk yield and longevity. In practice, genetic selection is based on many traits simultaneously, and realized selection responses depend on the selection intensity for the trait of interest. Determining the impact of a multi-trait selection strategy requires knowledge of genetic parameters (i.e. heritability, genetic correlation) and the values of individual traits in the breeding goal. Results presented in this study, therefore, provide a first step towards a better understanding of the potential of breeding to reduce GHG emissions from dairy production. Due to differences between feeding and breeding strategies under study, results of the strategies cannot be compared directly.

This study focused on the environmental impact of strategies related to GHG emissions. Dairy production, however, has an impact on the environment in other ways, such as eutrophication, acidification, and depletion of fossil energy and phosphorus sources. Including other environmental impact categories might change results.

5. Conclusion

Each feeding strategy evaluated in this study reduced GHG emissions per ton FPCM, but also reduced labor income of the farm family. Supplementation of nitrate resulted in the largest reduction in GHG emissions, but reducing the maturity stage of grass and grass silage resulted in lower costs, and a better cost-effectiveness. One genetic standard deviation improvement of milk yield and longevity resulted in a reduction in GHG emissions, while increasing labor income. The reduction in GHG emissions as well as the increase in labor income were more pronounced for milk yield than for longevity. Identification of strategies to reduce GHG emissions is a first step towards reducing the impact of dairy production in practice. Results indicate that a combination of different strategies is required to substantially reduce GHG emissions from dairy production.

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Can the environmental impact of livestock feed be reduced by using waste-fed housefly larvae?

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ABSTRACT

The livestock sector is searching for alternative protein sources, because of the expected increase in demand for animal products. Insects are such a protein source. Use of insects may reduce environmental impact as they have potential to turn organic waste into high quality insect-based feed products. The aim of this study was to explore the environmental impact of using common housefly larvae fed on organic waste as livestock feed. Data were obtained from a testing site, that is designing a rearing place of 20 tons of larvae meal per day. Results showed that larvae meal has a GWP of 770 g CO_2-eq, energy use of 9,329 MJ and land use of 32 m^2 per kg dry matter meal. Compared with soybean meal, larvae meal results in lower land use and a higher GWP due to its high energy use. To conclude, larvae production has potential to make livestock diets more sustainable.

Keywords: insects, house fly, larvae, LCA, environment

1. Introduction

A growing and more prosperous world population is demanding for more animal proteins, especially in developing countries (Godfray et al. 2010; Eisler et al. 2014). Livestock production, however, already and will continue to poses a severe pressure on the environment via their emissions to air, water and soil (Tilman et al. 2001; Steinfeld et al. 2006). Moreover, livestock also increasingly competes for scarce resources, such as land, water, and fossil energy (Steinfeld et al. 2006; Godfray et al. 2010). The current sector, for example, uses about 70% of the agricultural land (Steinfeld et al. 2006), mainly for pasture and production of feed crops. Land is scarce and expansion of the area for livestock production leads to deforestation in the tropics, i.e. 80% of new croplands are replacing forest, resulting in losses of ecosystem services, biodiversity and increased carbon emissions (Foley et al. 2007; Gibbs et al. 2010; Foley et al. 2011). Similarly, about 15% of the anthropogenic emissions of greenhouse gases result from livestock production (Gerber et al. 2013), mostly resulting from production and utilization of feed (De Vries and De Boer 2010).

So, there is an urgent need for efficient production of feed for livestock. Using insects as a protein source in livestock feed potentially enables such efficient production, i.e. by more efficient use of natural resources and low emissions to air, water and soil. According to a recent publication of the FAO insects as feed can emerge as an especially relevant issue in the twenty-first century (Van Huis et al. 2013).

Insects possess favorable characteristics: 1) They are highly nutritious and have value as a protein source for livestock (Veldkamp et al. 2012). Insect-based feed products, therefore, can replace conventional feed ingredients with a high environmental impact, like fishmeal and soybean meal (SBM). 2) Insects have a low feed conversion ratio and can be consumed as a whole (no residual materials i.e. no bones or feathers). 3) Insects may offer the possibility to reduce the environmental impact of livestock production. In contrast with feed cultivation, the production of insects is not necessarily land intensive (Van Huis et al. 2013). A further environmental benefit of the use of insects lays in their capability to turn organic waste streams, such as manure, household waste or formal food products, into high quality insect-based feed products (Veldkamp et al. 2012; Van Huis et al. 2013). By feeding waste-fed insects, livestock can be fed less food products that are directly edible by humans, thus reducing the competition for land. As an example, around 70% of the cereal grains used in developed countries is fed to livestock (Eisler et al. 2014). With a rather inefficient feed conversion ratio of livestock – for chicken 1.6, for pigs 2.5 and cattle 5.1 per kg dry matter feed/kg product (Šebek and Temme 2009) – more people could be supported from the same amount of land if they did not consume meat from livestock fed with cereals (Godfray et al. 2010).

Feeding waste-fed insects to livestock, therefore, might be an effective strategy to transform inedible waste streams for livestock and humans into high quality food products, such as meat, milk, and eggs. Already in 1970, Calvert et al. showed that housefly larvae (*Musca domestica L.*) can be used for biodegradation of chicken manure, while Ocio et al. (1979) showed that larvae can grow on municipal organic waste. As tons of manure and food waste are produced in western countries –according to the FAO one third of the food is never consumed (Gustavsson et al. 2011)- feeding insects organic waste streams seems a promising solution for the environment.

To our knowledge no study quantified the reduction of the environmental impact of livestock production by including waste-fed insects in livestock feed. The aim of this study, therefore, was to explore if the environmental impact of livestock production can be reduced by using larvae of the common housefly fed on organic waste streams as livestock feed. Environmental impacts included were land and energy use, and emission of greenhouse gases. Data were obtained from a commercially-exploited testing site that designs a rearing place for 20 tons of larvae meal per day. The larvae were fed with a substrate of poultry manure and food waste.

2. Methods

Life cycle assessment (LCA) is an internationally accepted and standardized holistic method (ISO14040 1997; ISO14041 1998; ISO14042 2000; ISO14043 2000) to evaluate the environmental impact during the entire production chain (Guinée et al. 2002; Bauman and Tillman 2004). An attributional LCA was performed to assess the environmental impact, as we aimed, to analyze the environmental impact of larvae meal in a status quo situation.

2.1. Goal and scope definition

Figure 1 illustrates the production chain of the larvae meal. The system consists of four stages: egg production, larvae production, substrate/feed production for larvae and processing of larvae in order to produce larvae meal. In the egg production stage, pupae are brought into a cage and will eclose into flies within 2 days. Feed of the flies consists of sugar, milk powder and egg powder. Flies are kept at a temperature of 25 degrees Celsius. Female flies start to lay eggs after 7 days in an oviposition substrate, consisting of milk powder, yeast, fiber, vegetable oil and vitamins. Drinking water is provided by a nozzle system and water is used for cleaning. The output of the egg production consists of eggs with the oviposition-substrate. The larvae production stages starts with mixing eggs with the oviposition-substrate with larvae-substrate. Subsequently, larvae are kept at a temperature of 27 degrees Celsius and are full grown after 5 days. The larvae-substrate consists of 195 ton food waste, 65 ton laying hen manure and 1 ton premix. Per 4 kilograms of substrate, one kilogram of larvae is produced. After harvesting the larvae, the cage is cleaned with water. Harvesting of the larvae is performed by shutting off the ventilation, which makes the larvae crawl to the surface of the substrate when oxygen levels drop. Per day 65 ton of live larvae are produced resulting in 20 ton of larvae meal with a dry matter (DM) content of 88%. Besides the larvae meal, 159 ton larvae manure is produced. The larvae manure is not a waste product as it can have different application e.g. as fertilizer. The environmental impact related to the larvae manure was not included in this study. Larvae meal was compared with fishmeal and SBM as both products are protein rich. The functional unit to calculate the impact of larvae meal is one kilogram DM of larvae meal.

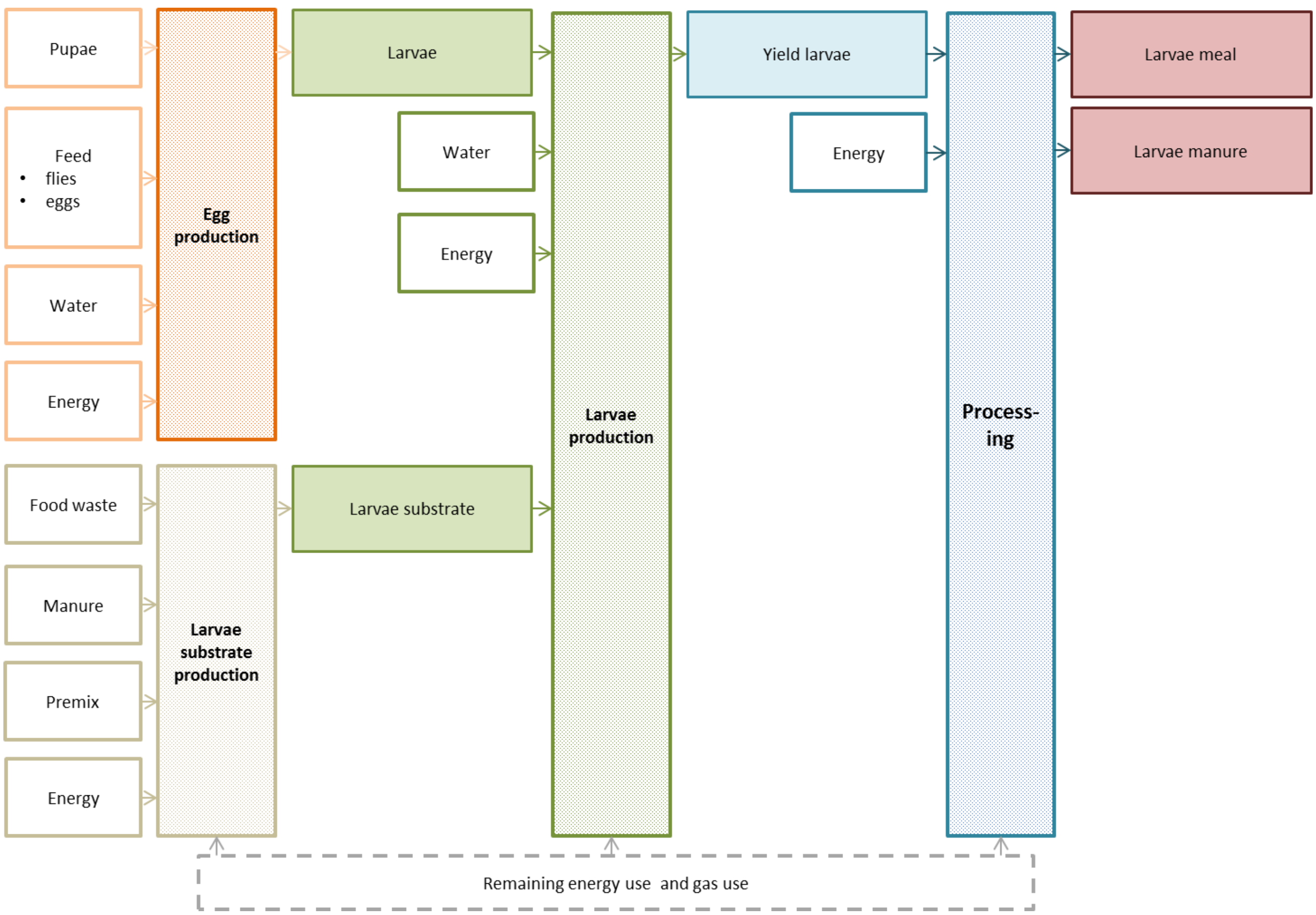

Figure 1. Stages in the production chain of larvae meal

2.2. Data collection

The model, based on experimental studies, is developed by four companies in the Netherlands (an animal nutrition company, Denkavit, two waste processing companies, AEB and SITA and an insect rearing company, Jagran). All data were provided by those four companies and a summary of the data is presented in table 1.

2.3. Measuring environmental impact

The impact categories greenhouse gas (GHG) emissions, energy use (EU) and land use (LU) were assessed. Emission of GHGs, EU and LU were assessed because the livestock sector contributes significantly to both climate change and LU worldwide (Steinfeld et al. 2006) and earlier results of Oonincx and De Boer (2012) demonstrated that insect production is related to high energy use. The following GHGs were included: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). These GHGs were summed up based on their equivalence weighing factors in terms of CO_2 (100 years' time horizon): i.e. 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al. 2007). Land use was expressed in m^2 per year per kg dm of larvae meal and energy use was expressed in MJ per kg dm of larvae meal. For both, parameters of GHGs, EU and LU and quantitative input data a sensitivity analyses was performed. In case of multifunctional processes, economic allocation was used. Economic allocation is the partitioning of the environmental impact between co-products based on the relative economic value of the outputs (Guinée et al. 2002).

A summary of the input and output data required to maintain the production of larvae meal together with the related impact factors is provided in table 1. The GWP, EU and LU related to the feed for the flies and egg substrate were based on Vellinga et al. (2013) for the production of feed ingredients and on EcoinventCentre

(2007) for the production of tap water. Environmental impact from production of feed ingredients included impacts from cultivation (e.g. fertilizers, pesticides, machinery, energy, emissions related to direct and indirect N_2O and CO_2 emissions from liming and urea fertilization) impacts from drying and processing, and impacts from transport up to the farm gate.

The emissions related to the substrate for the larvae were based on IPCC (2006). According to IPCC, emissions of methane from organic waste occur only after several months. As food waste was used for 4 days only during the larvae production process, we assumed that emissions from organic waste were negligible. During the handling and storage of laying hen manure CH_4 and direct and indirect N_2O were emitted. As there were no specific data available of the use of manure for insect rearing, we assumed emissions for using manure were equal to emissions emitted on a laying hen farm. For CH_4 a tier 2 approach was used based on country specific data of Coenen et al. (2013) and IPCC default values (an organic matter content of 0.35 kg per kg manure, maximum CH_4 producing potential of 0.34 m^3 CH_4 per kg organic matter and a methane conversion factor of 0.015). For direct N_2O emissions a tier 2 approach was used based on country specific data of Coenen et al. (2013) (0.8 kg N excretion per laying hen per year, 18.9 kg manure per laying hen per year and a default emission factor of 0.01). For indirect N_2O emissions a tier 1 approach was used based on IPCC default values (volatilisation 40% and an emission factor of 0.01). The GWP, EU and LU for transportation of food waste and manure over an average of 65 km per day were included, based on Eco-invent (2007). The GWP, EU and LU related to the production of electricity and gas were based on Eco-invent (2007). Electricity was assumed to be substituted with marginal Dutch electricity, i.e. 28% coal-based, 67% natural gas-based, and 5% wind-based electricity (EcoinventCentre 2007). The GWP, EU and LU related to SBM and fishmeal were based on Vellinga et al.(2013).

Table 1. Input data and related global warming potential (GWP), energy use (EU) and land use (LU) data for the environmental impact of producing one ton dry matter larvae meal.

Ingredients	**Unit**	**Amount** /ton (DM)	**GWP** (g CO_2)	**EU** (MJ)	**LU** (m^2)
Feed flies	kg	1	3,808	12.2	1.34
Substrate eggs	kg	17	1,351	3.9	0.34
Food waste	kg	11,079	11	0.2	0.00
Manure	kg	3,693	42	0.2	0.00
Premix	kg	57	1,362	3.9	0.34
Water	kg	10,309	0	0.0	0.00
Electricity	kWh	378	753	11.8	0.01
Gas	kWh	183	586	11.2	0.00

3. Results

Producing larvae meal resulted in a GWP of 770 kg CO_2-eq, an EU of 9,329 MJ and a LU of 32 m^2 per ton DM larvae meal. Figure 2 shows the GWP, EU and LU for different production phase including a sensitivity range of 30%. The egg production phase contain processes like feed and water use. The larvae production phase contains processes like water and substrate use. The electricity for the building contained the complete electricity use and gas for the building contained the complete gas use. The gas for drying the larvae is in this business model obtained from residual heat from a waste incineration plant in which the larvae production is situated. The largest part of the GWP was caused by the feed for the larvae (44%), whereas an additional 37% resulted from the use of electricity and 14% form the use of gas. Electricity and gas use, however, explained the majority of the EU (70%), whereas production of vitamins and minerals in larvae feed explained the majority of the LU.

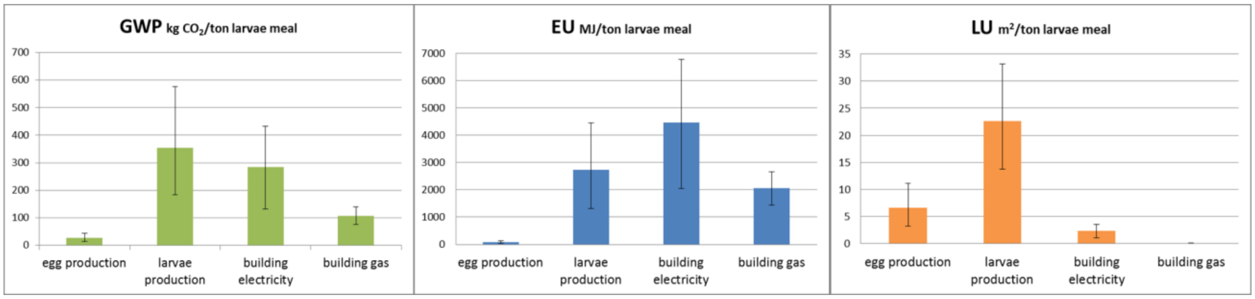

Figure 2. Global warming potential (GWP), energy use (EU) and land use (LU) of each phase of larvae meal per ton dry matter larvae meal, including a sensitivity range of 30%

To compare larvae meal with other protein rich ingredients a comparison of the nutrient content is of importance. Table 2 shows the nutrient content of larvae meal (based on analysis of the commercially-exploited testing site), fishmeal and SBM (CVB 2010).

Table 2. Nutrient content (%) of larvae meal, fishmeal and soybean meal (SBM)

	Larvae meal	SBM	Fishmeal
Dry matter	88.0	87.5	92.7
Crude protein	47.9	46.0	56.7
Fat	24.2	18.4	15.8
Lysine	32.6	28.5	43.1
Methionine	11.3	6.4	15.9

Figure 3 shows a comparison for the average GWP, EU and LU with other protein rich feed ingredients. The production of larvae meal and fishmeal results in high EU affecting the GWP but are not land intensive like SBM.

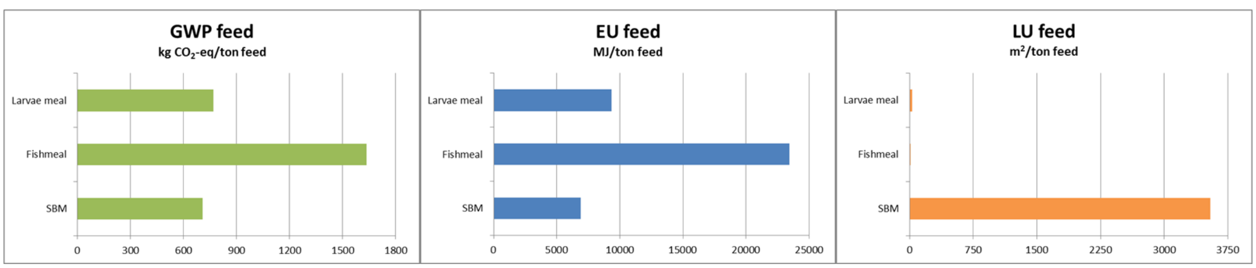

Figure 3. Comparison of global warming potential (GWP), energy use (EU) and land use (LU) of larvae meal, fishmeal and soybean meal (SBM) based on ton dry matter feed

4. Discussion

The results of this conference paper show the direct environmental impact of larvae meal production. Production of larvae meal, however, also resulted in indirect environmental consequences. The human food waste which was used to grow insects can be used for different application e.g. composting or production of biogas. The indirect environmental consequences of replacing another potential application of waste used for insects, therefore, should be considered as well. Just as the possible applications of the larvae manure. The larvae manure can, for example, be used as fertilizer or for biogas production. Further research on the indirect environmental impact is, therefore, required.

The range of GWP, EU and LU is based on a sensitivity analysis in which the values of GWP, EU and LU and quantitative input data were decreased and increased with 30%. Varying the values of emissions was required as some processes are uncertain. The most uncertain factor were the emissions related to the larvae substrate. Calculation of emissions were based on IPCC guidelines for manure and composting of food waste. We assumed that emissions of food waste were negligible as according to IPCC emissions of composting only occur after several months (IPCC 2006). Even though food waste is used for only 4 days, the circumstances for composting were favorable due to high temperatures and constant ploughing by the larvae. Un underestimation is, therefore, possible. Furthermore, IPCC calculations for manure were based on emissions related to the complete laying hen sector and not only for the storage of manure and, therefore, possibly resulting in an overestimation. To minimize the uncertainty experimental studies are required.

The nutrient content of larvae meal was determined on basis of only two samples. However, a literature review of Veldkamp et al. (2012) shows similar outcomes: larvae contain 43-68% protein and 4-32% fat on a dry matter basis. The protein content of insects is within the fishmeal and SBM range and its fat content is higher (Veldkamp et al. 2012).However, in vivo animal studies are required to determine the palatability, digestibility and other relevant characteristics of the larvae meal before a reliable comparison with fishmeal and SBM can be made.

In this study, we found that electricity use per kg DM of larvae meal was 4.46 MJ an gas use was 2.05 MJ. For mealworm production, a high electricity and gas use was also found. Oonincx and De Boer (2012) found an electricity use of 15.8 MJ and a gas use of 26.0 MJ per kg of DM mealworms. These values are thus higher than that of the production of housefly larvae, which is caused by a longer production cycle of mealworms (10 weeks instead of 5).

The production of larvae uses high amounts of energy due to the required ambient temperature. One should, however, take into account that the required energy use is an estimation and the bio-efficiency of the industrial process to acquire larvae meal is still advancing. However, the high cost price and institutional barriers (EU-legislation, health concerns etc.) are issues that should be overcome before an insect-based business model can be exploited (Veldkamp et al. 2012; Van Huis et al. 2013).

5. Conclusion

Energy use was the main contributor to the direct environmental impact of larvae meal production, however, the industrial process to acquire larvae meal is still advancing. Compared with fishmeal, larvae meal resulted in a lower GWP and EU and a similar LU. Compared with SBM larvae meal resulted in a higher GWP and EU but a lower LU. Two of the main production factors, land and fossil energy, are scarce. However, fossil energy can be replaced by more sustainable sources, i.e. solar- and wind energy, that reduces the GWP, while there is no practical solution for the scarcity of land. Therefore, we conclude that in the future larvae production has the potential to contribute as a more environmentally sustainable livestock feed.

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Environmental sustainability pathways based on a single raw material: European pilchard (*Sardina pilchardus*) in NW Spain

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ABSTRACT

European pilchard (*Sardina pilchardus*) is a commonly purchased species in Portugal and Spain, since it constitutes a cheap and healthy source of protein. Therefore, numerous final products are commercialized based on this same raw product. The current study presents a cross-product environmental analysis using Life Cycle Assessment (LCA) for three different final consumer-purchased products based on European pilchard landings in Galicia (NW Spain): canned pilchards, fresh pilchards and European hake caught in the Northern Stock using pilchard bait. Furthermore, the entire life cycle of the products were considered, including a series of different cooking methods for each final product. The midpoint and endpoint results obtained showed important differences in the final environmental impacts between the products, with the different cooking methods also appearing to be a crucial source of uncertainty. The results are analyzed regarding relevant limitations, uncertainties and consumer and policy implications through the use of Mixing Triangles.

Keywords: canning industry, consumption patterns, LCA, Mixing Triangle, seafood

1. Introduction

Iberian countries (i.e., Portugal and Spain) are among the few countries in the world in which protein supply for human consumption associated with the intake of seafood products represents over 20% of the daily protein intake of animal origin (FAO 2012). More specifically, European pilchard (*Sardina pilchardus*) constitutes an attractive seafood product in Spanish and Portuguese households. For instance, the annual per capita consumption of pilchard in Spain is currently around 2.1 kg, only behind tuna and hake products in terms of seafood products (Martín-Cerdeño 2010). Pilchards are usually consumed fresh in the summer months (June to early October) due to their higher levels of fat in this period, improving their taste and aroma (Zlatanos and Laskaridis 2007). During the rest of the year, however, most of the landed pilchard is used mainly for the canning industry or as bait in demersal fisheries. In 2009 a total of nearly 30,000 metric tons of pilchards were canned in Spanish territory, of which approximately 80% was canned in Galicia, a region in NW Spain known for its seafood industry (ANFACO 2011).

Independently of the current overexploitation of the European pilchard stock off the coast of Portugal and NW Spain or the natural fluctuations that this species presents from one year to another, the environmental impacts linked to the operation of the fishing fleet that captures pilchard has historically been ignored. However, the landing, processing and consumption of seafood products has shown to be a relevant source of greenhouse gas (GHG) emissions, due to the intensive energy use in the fishing and canning sectors (Hospido et al. 2006; Vázquez-Rowe et al. 2013). Therefore, an increasingly used methodology to monitor in an integrated manner a range of environmental impacts linked to anthropogenic activities, including fishing, is Life Cycle Assessment – LCA (ISO 2006a). In fact, this method has been used in several recent publications to understand the main environmental dimensions that are affected by fishing activities and their derived supply chains (Vázquez-Rowe et al. 2012).

The main aim of the current study was to understand the environmental impacts using an LCA perspective associated with the entire life-cycle of a selection of three seafood products available for Spanish consumers in supermarkets and other retailing stores derived from the landing of European pilchard. The latter raw material was monitored based on landings performed by Galician purse seiners. However, beyond these particular landings, a cross-product analysis was performed based on an equal amount of protein supply at the plate of the consumer, with the aim of analyzing the variable environmental impacts linked to different processing and/or consumption routes for one single raw product.

The described goals intend to provide support for decision-making at a company level throughout the supply chain, as well as a starting point for the evaluation of integrated policies for the reduction of environmental impacts in the food sector. Finally, the results will also guide consumers in terms of responsible consumption of seafood products (González et al. 2011).

2. Methods

2.1. Goal and scope definition

The main function of the production system is that of nourishing human communities with a specific amount of protein content (an essential part of human diets) embedded in seafood final products (Pimentel and Pimentel 2003). However, in this specific study a comparison is provided between three different final products at the retailing center for the consumers to purchase, but deriving from one single raw product: European pilchard. Hence, the functional unit (FU) selected was fixed as the amount of protein available (17.26 g) in *Scenario A*, that is, a can of pilchards in olive oil (85.0 g) produced by a Galician canning factory. For the alternative scenarios, the same amount of protein available for the consumer was considered. Therefore, in *Scenario B* a total of 137 g of fresh round pilchard were assumed ready for intake. Similarly, in *Scenario C* the consumers' final intake is 183 g of European hake landed by Galician long liners. However, in the latter scenario it should be noted that this hake is captured using pilchard as bait (75.2 g per FU), which justifies its comparability with the previous scenarios.

2.2. System description

Figure 1 presents a schematic representation of the different subsystems included in the life-cycle of each scenario. Subsystem I comprises the fishing activities to capture and land pilchard and, therefore, is common to all scenarios together with Subsystem II, which includes the auction and port activities. In contrast, Subsystem III only applies to Scenarios *A* and *C*, since *Scenario B* does not include a processing stage (the pilchard is consumed fresh). For the former, Subsystem III comprises the entire processing of the canned pilchard, including the reception and storage of the raw pilchard, cutting and canning of the cleaned pilchard, cooking and the addition of more ingredients (i.e. olive oil and salt), sterilization, packaging and, finally, the distribution of the processed product to the regional distribution center. For *Scenario C*, however, this subsystem is associated with the processing, freezing and distribution of pilchard in the form of bait to supply the Galician long liners, as well as the fishing stage, auction and port activities for the hake landings. Subsystem IV comprises the wholesaling and retailing of the different products, which are thereafter consumed in Galician households (Subsystem V). Moreover, Subsystem V also includes the end-of-life (EOL) activities of the different materials used throughout the supply chain that are disposed of by the consumer (e.g., tins, packaging, organic residues…). Finally, Subsystem VI deals with the human excretion phase based on the assumptions available in Muñoz et al. (2008).

2.3. Data acquisition

Data for the European pilchard fishery were retrieved from a previous study developed by Vázquez-Rowe et al. (2010) that analyzed the Galician purse seining fleet from an LCA approach. Landing and auction activities were inventoried based on data disclosed by the port of Vigo (Autoridad Portuaria de Vigo, personal communication, September 2010). For the processing stage, data for the canning factory were obtained from a canning factory located in *A Coruña* province. This factory destines 95% of its production efforts to deliver canned pilchard. For the baiting data, however, data were delivered by a bating SME located in *Lugo* province. In addition, European hake landings were obtained from a study undergone by Vázquez-Rowe et al. (2011) that analyzed hake landings by Spanish vessels in the North Atlantic. Finally, the remaining inventory data were retrieved from a wide range of bibliographical sources, including the ecoinvent® database, from which background data for most inventory items were modelled (Frischknecht et al. 2007).

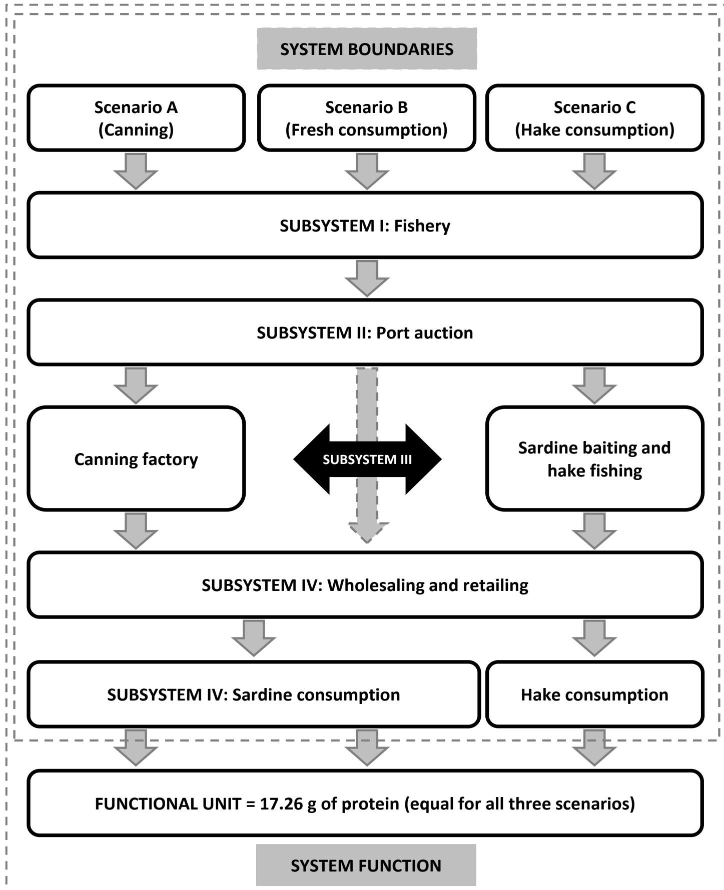

Figure 1. Schematic representation of the analyzed production systems.

2.4. Life Cycle Inventory

The Life Cycle Inventory (LCI) consists of the organization and structuring of the data collected to perform the LCA (ISO 2006b). Table 1 presents a summarized LCI for the three subsystems.

2.5. Allocation and other assumptions

Two different allocation approaches were considered throughout the supply chains analyzed. On the one hand, allocation was identified as been necessary in Subsystem I, since pilchard is captured in a multispecies fishery together with Atlantic mackerel or horse mackerel (Vázquez-Rowe et al. 2010). As recommended by ISO 14044 and the PAS 2050 appendix for seafood products, mass allocation was selected with the aim of accounting for the physical relationship between co-products (ISO 2006b; BSI 2012). On the other hand, in *Scenario A* an important intermediary residue is generated in the canning factory: the portion of the life weight of pilchards that is not used in the cans (141 g per FU). For this particular co-product scenario, three different allocation approaches were used in the three alternative situations considered for *Scenario A*: i) economic allocation based on economic revenues in the canning plant; ii) economic allocation considering economic savings in the factory; iii) energy allocation, which takes into consideration the final amount of protein that the residues provide to humans diets when used for reduction and aquaculture feeding.

Table 1. Summarized Life Cycle Inventory (LCI) for the production systems analyzed. Data referred to the fixed functional unit.

	Unit	Scenario A			Scenario B		Scenario C	
		Economic revenue allocation	**Economic savings allocation**	**Energy allocation**	**Fried pilchards**	**Grilled pilchards**	**Fried hake**	**Boiled hake**
INPUTS								
SUBSYSTEM I								
Diesel	g	39.89	33.91	29.14	24.13	24.13	13.24	13.24
Ice production	g	72.76	61.85	53.14	44.01	44.01	24.14	24.14
Steel	g	0.61	0.52	0.45	0.37	0.37	0.20	0.20
Seine net	g	2.31	1.97	1.69	1.40	1.40	0.77	0.77
SUBSYSTEM II								
Pilchard (live weight)	g	226.7	192.7	165.6	137.1	137.1	75.20	75.20
Polystyrene	mg	319.8	271.8	233.6	193.4	193.4	106.1	106.1
Electricity	kWh	3.51E-3	2.98E-3	2.56E-3	2.13E-3	2.13E-3	1.17E-3	1.17E-3
SUBSYSTEM III								
Diesel (hake fishery)	g	--	--	--	--	--	238.9	238.9
Light fuel oil (canning)	g	276.8	276.8	276.8	--	--	--	--
Olive oil (canning)	g	35.00	35.00	35.00	--	--	--	--
SUBSYSTEM IV								
Polyethylene	g	0.10	0.10	0.10	--	--	3.55	3.55
Electricity	kWh	0.01	0.01	0.01	0.02	0.02	0.03	0.03
Lorry transport	tkm	0.30	0.30	0.30	0.07	0.07	0.11	0.11
Van transport	kgkm	12.45	12.45	12.45	7.34	7.34	10.13	10.13
SUBSYSTEM V								
Olive oil	mL	--	--	--	9.60	--	13.73	--
Salt	g	--	--	--	0.66	0.66	0.46	0.46
Flour	g	--	--	--	4.66	--	4.58	--
Electricity (cooking)	kWh	--	--	--	0.52	--	0.14	0.12
SUBSYSTEM VI								
Tissue paper	mg	653.9	653.9	653.9	746.3	682.9	835.5	768.9
Water	L	2.06	2.06	2.06	2.35	2.15	2.63	2.42
Soap	mg	544.9	544.9	544.9	621.9	569.1	696.3	640.8
Ferric chloride	mg	0.57	0.57	0.57	0.66	0.60	0.72	0.68
Ferric sulfate	mg	0.42	0.42	0.42	0.48	0.44	0.52	0.49

2.6. Life Cycle Impact Assessment

Results computation was performed using the SimaPro software, version 7.3 (Goedkoop et al. 2010). The selected assessment method to calculate the final midpoint and endpoint results was ReCiPe (Goedkoop et al. 2009). For the endpoint results the hierarchist approach was selected.

3. Results

The total endpoint single score environmental impacts identified in *Scenario A* ranged from 0.52 to 0.58 Pt depending on which of the three residue allocation perspectives was selected, representing a maximum variability of only 11%. The canning stage (Subsystem III) accounted for 91% of the total endpoint impacts, while the fishery (4%) and the remaining subsystems (6%) only accounted for a minor part of the overall impacts. *Scenario B*, in contrast, was identified as the scenario with lowest environmental impacts per FU, 90%-95% lower than those found for *Scenario A* and 70%-83% lower than those for *Scenario C*. Nevertheless, important differences were detected depending on the way in which the fresh pilchards had been cooked in the consumer stage. More specifically, the overall environmental impacts of preparing pilchards on a barbeque (i.e., grilled) were quantified at 2.93E-2 Pt, 45% lower than those linked to the consumption of fried pilchards. In this scenario the main contributor to the overall environmental impact was the fishery stage (45%), followed by the consumption subsystem (24%) and wholesaling and retailing (15%) whenever the pilchards are grilled. However, if the pilchards are fried the consumption stage becomes the main contributor (57%), followed by the fishery stage (25%). Finally, the impacts related to *Scenario C* add up to 1.80E-1 Pt if the hake is fried or 1.75E-1 if the hake is boiled. Impacts in this scenario are dominated by the fuel use intensity of the long lining fleet in Subsystem III and, to a lesser extent, to the fuel combustion in the purse seining fleet and in terrestrial transport.

The impact categories that dominate the majority of the processes analyzed were fossil depletion, climate change and particulate matter formation, demonstrating the high reliance on fossil fuels of the processes analyzed, independently of the scenario. Midpoint results for the three scenarios are shown in Table 2.

Table 2. Total environmental impacts per scenario reported per functional unit (ReCiPe midpoint).

Impact categories	Unit	Scenario A			Scenario B		Scenario C	
		Economic revenue allocation	Economic savings allocation	Energy allocation	Fried pilchards	Grilled pilchards	Fried hake	Boiled hake
Climate change	kg CO_2 eq	3.36	3.09	2.88	4.71E-1	2.06E-1	1.44	1.38
Ozone depletion	kg CFC-11 eq	7.48E-7	7.32E-7	7.19E-7	2.01E-7	1.75E-7	6.65E-6	6.64E-6
Human toxicity	kg 1,4-DB eq	1.71	1.54	1.40	1.73E-1	9.89E-2	1.80E-1	1.39E-1
Photochemical oxidant formation	kg NMVOC	1.78E-2	1.69E-2	1.62E-2	3.57E-3	2.74E-3	2.09E-2	2.06E-2
Particulate matter formation	kg PM_{10} eq	1.20E-2	1.08E-2	9.83E-3	1.22E-3	8.61E-4	5.87E-3	5.72E-3
Ionizing radiation	kg U235 eq	9.18E-1	8.56E-1	8.06E-1	1.57E-1	1.62E-2	1.07E-1	7.33E-2
Terrestrial acidification	kg SO_2 eq	1.98E-1	1.85E-2	1.75E-2	3.36E-3	1.86E-3	1.53E-2	1.48E-2
Freshwater eutrophication	kg P eq	2.36E-3	2.23E-3	2.13E-3	4.00E-4	2.31E-5	2.26E-4	6.73E-5
Marine eutrophication	kg N eq	1.94E-3	1.91E-2	1.89E-2	2.87E-3	9-07E-4	3.53E-3	9.14E-4
Terrestrial ecotoxicity	kg 1,4-DB eq	7.60E-2	7.59E-2	7.59E-2	-7.21E-4	5.42E-4	-1.81E-3	2.06E-4
Freshwater ecotoxicity	kg 1,4-DB eq	1.41E-1	1.23E-1	1.08E-1	2.78E-2	1.75E-3	6.68E-3	3.58E-3
Marine ecotoxicity	kg 1,4-DB eq	2.02E-1	1.82E-1	1.67E-1	6.06E-3	2.61E-3	4.97E-2	4.87E-4
Agricultural land occupation	m^2a	3.94	3.93	3.92	3.02E-2	1.06E-1	2.36E-2	3.50E-2
Urban land occupation	m^2a	7.57E-2	7.19E-2	6.88E-2	2.78E-3	1.95E-3	3.72E-3	3.15E-3
Natural land transformation	m^2	5.92E-3	5.89E-3	5.85E-3	1.10E-4	7.83E-5	5.13E-4	5.08E-4
Water depletion	m^3	4.89E-1	4.87E-1	4.85E-1	1.85E-1	3.34E-3	2.64E-2	5.40E-3
Mineral depletion	kg Fe eq	6.10	5.20	4.49	3.33E-2	3.99E-2	2.57E-2	1.10E-2
Fossil depletion	kg oil eq	1.36	1.28	1.22	1.27E-1	5.54E-2	3.90E-1	3.78E-1

4. Discussion

Environmental impacts in *Scenario A* are strongly influenced by the operations occurring in the canning stage (Subsystem III). The combustion of fuel-oil, the use of packaging materials or the impacts related to the life-cycle of olive oil are all important contributors in this stage. However, the production and use of tin for canning is the main source of environmental impacts, especially in terms of climate change and fossil depletion. Tin impacts, that were modelled according to the ecoinvent® database, are known for the high degree of uncertainty (Classen et al. 2007). Having said this, in the current study these uncertainties were minimized by including recent recycling rates for tin in Spain. Therefore, it is quite obvious, as pointed out in previous studies (Hospido et al. 2006), that a change in the container for canned pilchard could be a convenient improvement action, as long as this measure does not imply a burden in sales due to consumer preferences (Calvo 2013).

The use of olive oil in the cans also presented high environmental impacts, although in this case the impacts were fairly evenly distributed between different stages of production (fertilizers, pesticides, refining, etc…). Hence, integrated minimization of impacts schemes will be necessary in the olive oil producing sector to reduce impacts in this particular operation (Iraldo et al. 2014). Finally, the fuelling of the fishing vessels, especially in Scenario C, in which the long liners constitute an important source of environmental impacts, was also identified as a main carrier of environmental burdens. Consequently, appropriate fuel optimization schemes or changes in the energy carrier of the vessels should be analyzed (Bengtsson et al. 2012).

Comparison for the different supply chains monitored in this study was performed with care, since there are some methodological issues that must be taken into consideration. For instance, there are a series of characteristics, such as the shelf life of the products, or the food waste they generate throughout the chains that can vary enormously. However, the lack of data and the assumption that the supply chains were centered in Galicia, a coastal region in which the arrival of fresh fish is done on a daily basis (unlike other landlocked areas in which the shelf life of fresh seafood is reduced considerably) manages to minimize the effect of this issue. Figure 2 shows the direct endpoint single score environmental impacts for each of the processes analyzed, showing a clear dominance of *Scenario A* (all three alternatives) in terms of overall environmental burdens. Nevertheless, it should be noted that the methodological assumptions regarding allocation in the canning phase imply considerable final impacts, although not sufficient to reduce the impact of canned pilchard to the level of Scenarios *B* or *C*. Consequently, it appears that the consumption of the three main ways in which the pilchard raw material is finally consumed by the general public presents three completely different patterns in terms of environmental burdens: i) *Scenario A* presents a high energy use in the processing stage, due to the use of different packaging material -namely tin- as well as the use of electricity and logistics; ii) *Scenario B* presents a very low environmental impact, which is in line with the fresh consumption of other small-pelagic fish (Ramos et al. 2011; Avadí et al. 2014); and iii) *Scenario C* presents a high energy intensiveness, which implies that the final environmental profile of consuming hake is among the highest in the literature (Vázquez-Rowe et al. 2012; Avadí and Fréon 2013).

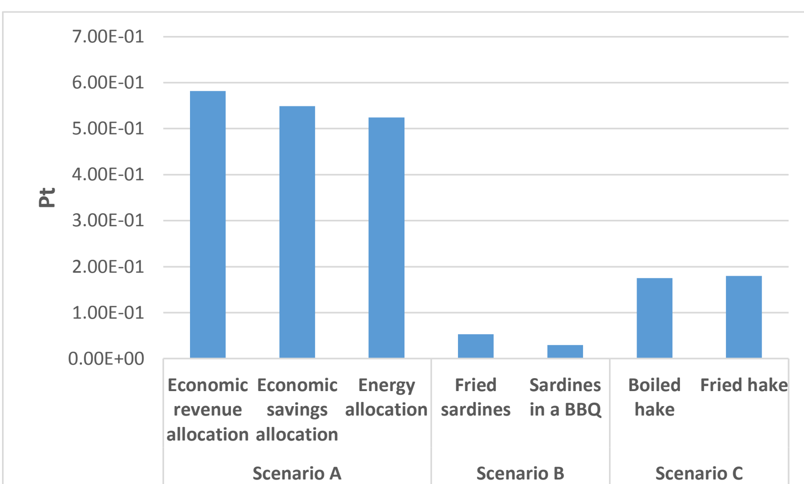

Figure 2. Endpoint single score environmental impact results for the selected scenarios (assessment method: ReCiPe endpoint H/H).

While the results when analyzed in subsystems do not show major surprises in terms of environmental impacts as compared to previous studies available in the literature, to the best of our knowledge the current study is the first attempt to follow the entire life-cycle of seafood products, including consumption, EOL of wastes and human excretion providing an extended view of a variety of different supply chains based on pilchard extraction. Interestingly, the cooking, consumption and EOL stage of the different products in households did not appear to be a major factor affecting the comparability of the three scenarios. However, whenever each scenario is analyzed depending on cooking options, the preparation of fried fresh hake or pilchard showed increased environmental impacts than when other cooking options were considered, such as grilling or boiling, demonstrating that consumer behavior can also have an important impact on the final profile of the products (Vázquez-Rowe et al. 2013b).

Having said this, it is important to highlight that the comparison between cooking methods was done with a fixed weighting of damage categories. Hence, the ReCiPe hierarchist single score values reported in this study imply a 40% weighting of ecosystem damage categories (land use and toxicity categories, together with acidification and climate change), 30% for human health categories (which include climate change (partially, ozone depletion or human toxicity) and 30% for resources (mineral and fossil depletion). However, the use of the MIXTRI 2.0 model, developed by Doka (2011) was selected in order to analyze how different weighting possibilities may lead to variable final results (Hofstetter et al. 1999). In fact, given the high use of energy in the fisheries and transportation stages of the systems analyzed, as well as the use of important amounts of tin in the canned pilchard processing phase, it is arguable that the resources damage category should be weighted higher. Hence, Figure 3, using an uncertainty interval of 35% due to the numerous assumptions considered and described in section 2, represents the Mixing Triangles for Scenarios *B* and *C*. In the first place, the results for *Scenario A* show dominance for the energy allocation perspective. While this is barely surprising due to the allocation assumptions included in each perspective, implying that lower values of inputs and outputs are allocated to most of the processing stage in the energy perspective, the results do confirm the importance of methodological assumptions. However, these results were not represented in Figure 3 since they do not represent different consumption pattern options. Secondly, for fresh consumption of pilchard, the triangle showed complete dominance of grilled pilchard despite the high uncertainty range considered. Finally, for *Scenario C* the results showed that the relative dominance of boiled hake is highly constrained by the different sources of uncertainty underlying the calculations. While the results presented in the Mixing Triangles do not provide any further findings, they do demonstrate the strong dominance in environmental impacts for certain consumption patterns of one single raw product and consumer behavior trends. Consequently, beyond the relevant findings in terms of improvement actions that can be undergone in the different production systems, these results could be potentially used to drive consumer awareness policies by policy-makers with the aim of improving the environmental sustainability of human diets.

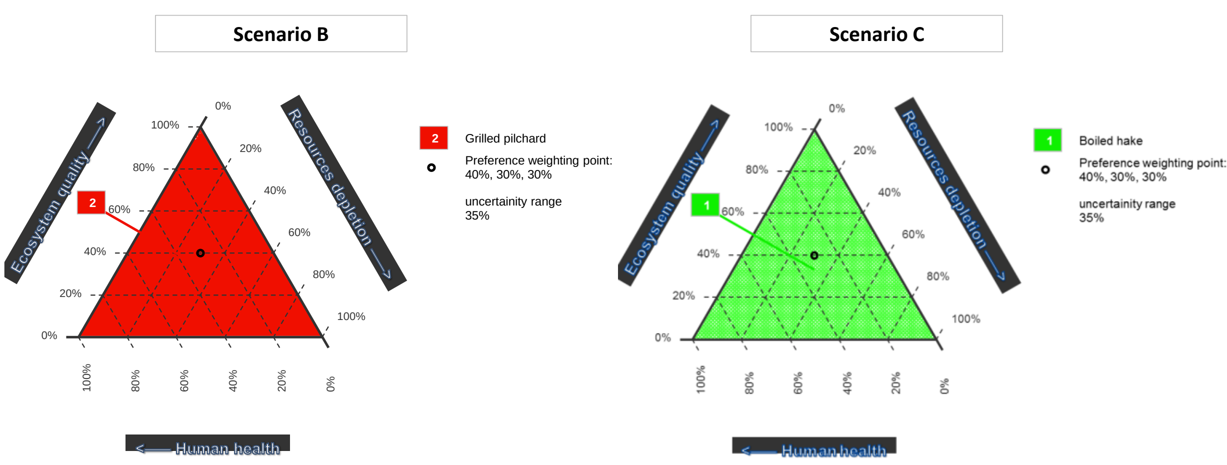

Figure 3. Weighting triangle matrix for different spreading techniques in Scenarios *B* and *C*.

5. Conclusion

The results presented in this study suggest that consumer choices in terms of how to intake a specific raw material can have important consequences on environmental impacts. For instance, the consumption of fresh round pilchard implied environmental impacts up to 95% lower than if the pilchard is consumed in cans. In addition, as pointed out in a previous study by Vázquez-Rowe et al. (2013b), cooking methods can also determine to a great extent the amount of environmental impact produced by the analyzed system.

Therefore, we argue that the analysis of the entire life-cycle of seafood products can provide important additional information for policy-makers, retailers, processors and consumers regarding sustainable seafood consumption patterns. In other words, while the extraction of a specific raw material can drive the main environmental profile of a final product, the on-land stages can also have an important role when it comes to mitigating or augmenting the final footprint of these products.

Nevertheless, future research should focus on analyzing these patterns at a broader scale -including the complex wholesaling and retailing networks that occur in the seafood sector- as a way to steer upcoming seafood and general food policies, as well as increasing awareness among consumers regarding the important role that their decision-making can have on the final environmental impact of seafood products.

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Dynamic Life Cycle Assessment of the *Ribeiro* wine appellation (NW Spain) in the period 1989-2009

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ABSTRACT

Land use changes (LUCs) are an important source of environmental changes in production systems, especially in the agricultural sector, where LUCs have been found to be a relevant factor to take into consideration when analyzing greenhouse gas (GHG) emissions. The viticulture subsector, as part of a broader agricultural sector, is not alien to the problematic of GHG emissions and climate change. Spain, as one of the main producers of wine worldwide, but also due to the important legislative and productive changes that have occurred ever since it joined the European Union, plays an important role in the analysis of how LUCs linked to the viticulture sector have effects on the environment. Therefore, in the current study we examined the LUCs that have occurred in the Ribeiro appellation in NW Spain between 1989 and 2009. For this, GIS was used to map the gradual dynamic changes on an annual basis of the areas used for wine production. Thereafter, the different land uses that substituted or were substituted by vineyards were identified in order to calculate the carbon storage and carbon emission dynamics based on the IPCC guidelines. Finally, operational activities linked to viticulture, including changes in technology over time, were added to the model in order to obtain a broad picture of entire impact of viticulture in terms of GHG emissions. The results present an interesting pattern, with GHG emissions linked to LUCs steadily decreasing in the timeframe analyzed. Similarly, the improvement of machinery, the reduction on fossil dependency in the Spanish electricity mix and the stricter use of specific standards for the use of plant protection agents also contributed to a gradual decrease in GHG emissions per functional unit (i.e. 1 ha of cultivated vineyards). These results are aimed at providing the appellation and other appellations throughout with environmentally relevant information regarding how different factors influencing their change through time can be analyzed to give support in policy making and decision making at a business level.

Keywords: appellation, dynamic LCA, land use changes, Life Cycle Assessment, viticulture.

1. Introduction

Wine production in Spain and other Mediterranean countries constitutes an important source of income in many rural areas. In fact, Spain is the country in the world with the highest surface area destined to wine production, even though the final gross productivity is still higher in Italy and France. In addition, the increasing interest in wine tasting worldwide, together with the limitations in wine-growing enforced by the Common Agricultural Policy of the European Union, have produced important changes in the sector since the early 1990s. In the first place, an increasing number of wine producing regions have clustered in appellations, in order to increase the visibility of the wine produced abroad, as well as to create a series of standards of production. This has led to higher level of technological improvements in the appellations, as compared to subsistence practices used until the mid-1980s. Secondly, most rural areas in Spain have suffered important demographic losses in the past 30 years, due to a wide range of factors, which include urbanization and the impact of the new European agricultural policies. Finally, changes in the wine consumption market, which is now targeted to a more educated wine taster, in which the wine product is no longer a companion of a meal, but a quality product in itself, have also led wine-growers to change the markets they serve and the way in which wine products are presented.

All these factors that have been briefly described have derived in important changes in the landscape of numerous appellations across Spain, with a gradual decrease in surface area destined to wine making, but with more intensive practices and with a higher level of appellation control in terms of operational practices (e.g. use of plant protection agents or application of fertilizers) on the individual wine-growers. In addition, these changes in landscape have also created changes in two important spheres. On the one hand, on the land use changes (LUCs), an issue that has been identified by the IPCC as an important generator or inhibitor of greenhouse gas (GHG) emissions, depending on the direction of the LUC. Hence, the substitution of agricultural land by forestry is most likely to create conditions for carbon storage, not only from an aerial perspective in the vegetation, but also due to the enhancement of the storage capacity of the soil (Williams et al. 2011). On the other hand, technological improvements and changes in the use of materials in the viticulture sector have also occurred in the past two decades. For instance, the trellis, which was traditionally constructed with wood, has been gradually substituted by other materials through time, such as granite slabs, and more recently, slate slabs or galvanized iron.

Based on these on-going trends in the sector, the main aim of this study is to understand from a timeline approach how these changes are affecting the environmental profile of wine produced in a selected appellation in NW Spain: *Ribiero*, located in the autonomous community of Galicia. The selected method for this analysis was Life Cycle Assessment (LCA), an internationally standardized assessment method that provides, based on detailed inventories with a life-cycle approach, an integrated analysis of different environmental dimensions, such as climate change, toxicity or use of resources among others (ISO 2006a). Therefore, LCA is used to monitor the changes in the environmental profile of *Ribeiro* wine from a timeline perspective from 1989 to 2009, with the aid of GIS mapping and the IPCC guidelines, in order to understand the evolution of its environmental burdens and to analyze the implications it may have in future policy making and decision making at a business level.

2. Methods

2.1. Goal and scope of the study

The main goal of this study was to understand the environmental profile of wine production in the *Ribeiro* appellation in the period 1989-2009. The selected functional unit (FU) for this case study was fixed as 5380 ha of land. This choice of FU is based on the fact that this was the entire land surface that was destined to wine production in the period 1989-2010. In other words, all land used for viticulture in this period was included within the system boundaries, regardless of the fact that either this land may have changed land use during this period, or that at the beginning of the period (1989) the land had a different use other than viticulture. This choice of a land FU is based on the lack of data regarding the yield on a temporal basis. Moreover, even if it were available, it would be subject to interannual changes due to natural fluctuations in yield. Therefore, the FU that was finally used serves as an adequate proxy to determine the changes in environmental impact in the appellation in the past two decades.

The system boundaries of the study included, as aforementioned, the entire surface destined to viticulture in the appellation, regardless of any LUC that may have occurred in the timeline under analysis. Based on this geographical boundary, that includes a total of 5380 ha, the biogenic carbon interactions on-going on an annual basis were monitored based on the IPCC guidelines for monitoring carbon release and/or storage due to LUCs. In addition, the entire life-cycle of viticulture activities, such as the production and use of fertilizers and plant protection agents, the use of trellis, machinery and operational activities in the vineyards, as well as the final harvesting activities up to the gate of the vinification plant, were included within the boundaries.

2.2. Data acquisition and IPCC guidelines

Data were obtained from a wide range of sources. In the first place, data on LUCs changes in the appellation were obtained from land use maps purchased from the Spanish Ministry for the Environment. These maps were only available for the period spanning from 1989 to 2009 and, therefore, guided the final time delimitation (see Table 1). Secondly, based on these LUCs, the IPCC guidelines for LUCs were taken into consideration to monitor which land use dynamics had a positive impact on carbon storage, or whether these changes were actually contributing to carbon emissions, as depicted in Table 2 (IPCC 2006). In addition, data for operational activities of vineyards in the appellation were retrieved from previous LCA studies analyzing the appellation (Vázquez-Rowe et al. 2012; 2013; Villanueva-Rey et al. 2013). Finally, data on trellis appeared to suffer major changes in the past two decades. Therefore, the appellation authorities provided guidance regarding changes in trellis materials in vineyards over time (DO Ribeiro, 2014, personal communication).

Table 1. Land use changes for selected years in the *Ribeiro* appellation. All values presented in hectares.

	1989	1993	1997	2001	2005	2009
Forest land	535	428	321	214	107	0
Forest land (transition)	0	323	645	968	1290	1613
Vineyards – Forest	0	323	645	968	1290	1613
Vineyards	4448	3981	3513	3046	2578	2111
Vineyards (transition)	0	186	373	559	746	932
Forest – Vineyards	0	107	214	321	428	535
Meadows – Vineyards	0	16	32	49	65	81
Other crops - Vineyards	0	31	62	94	125	156
Other land – Vineyards	0	32	64	96	128	160
Meadows	81	65	49	32	16	0
Meadows (transition)	0	20	40	60	80	100
Vineyards – Meadows	0	20	40	60	80	100
Other crops	156	125	94	62	31	0
Other crops (transition)	0	35	70	105	140	175
Vineyards-Other crops	0	35	70	105	140	175
Fallow land	160	128	96	64	32	0
Fallow land (transition)	0	77	154	230	307	384
Vineyards - Fallow land	0	77	154	230	307	384
Wetlands	0	0	0	0	0	0
Wetlands (transition)	0	13	26	39	52	65
Vineyards – Wetlands	0	13	26	39	52	65
TOTAL	**5380**	**5380**	**5380**	**5380**	**5380**	**5380**

Table 2. Land use changes for selected years in the *Ribeiro* appellation. All values presented in hectares.

Land use		Emissions/capture of CO_2	Emissions/capture other GHG	Total t CO_2eq
Initial land use	Transition to…	Metric tons of CO_2	t CO_2 eq	
Forest land	Forest land	-24.35	0.13	-24.22
Vineyards	Forest land	-35.53	0.13	-35.40
Vineyards	Vineyards	18.40	0.57	18.97
Forest land	Vineyards	241.61	0.99	242.60
Meadows	Meadows	0.00	0.00	0.00
Vineyards	Meadows	18.27	0.00	18.27
Meadows	Vineyards	0.47	0.75	1.22
Fallow land	Fallow land	0	0.00	0
Vineyards	Fallow land	19.28	0.00	19.28
Fallow land	Vineyards	-5.74	0.62	-5.12
Vineyards	Wetlands	21.83	5.69	27.52
Other crops	Other crops	0	0.50	0.50
Vineyards	Other crops	-2.41	0. 50	-1.91
Other crops	Vineyards	-4.76	0.72	-4.04

2.3. Life Cycle Inventory

The Life Cycle Inventory (LCI), as explained briefly in the previous section included LCI data for the appellation used in prior studies (Vázquez-Rowe et al. 2012; 2013; Villanueva-Rey et al. 2013). However, it was important to adapt the operational activities to the entire extent of the period analyzed in two different steps. On the one hand, contacts were made with stakeholders in the appellation in order to understand the operational changes, including machinery, use of fertilizers and plant protection agents. Moreover, specific data were disclosed by the appellation authorities linked to historical data on trellis and use of pesticides. On the other hand, once these data were structured correctly, and inserted in the software selected (i.e., SimaPro v8.0), background processes supporting these data were adapted to the different years of assessment. For this, the electricity mix for Spain was modelled for all years of analysis and inserted into the software, in order to link each production process (i.e., production of fertilizers, pesticides, etc.) to the most appropriate electric profile (see Table 3). In addition, similar processes were done regarding machinery and the use of fossil fuels.

Table 3. Electricity mix profile for Spain in selected year of assessment. Data reported per 1 kWh.

	1989	**1993**	**1997**	**2001**	**2005**	**2009**
Electricity, hard coal, at power plant	0.3102	0.3649	0.2806	0.2307	0.2403	0.1255
Electricity, lignite, at power plant	0.0772	0.0847	0.1532	0.096	0.0344	0.0063
Electricity, oil, at power plant	0.0743	0.012	0.0503	0.1119	0.0833	0.06339
Electricity, natural gas, at power plant	0.0118	0.0058	0.0384	0.1059	0.2695	0.3853
Electricity, industrial gas, at power plant	0	0	0	0	0	0
Electricity, hydropower, at power plant	0.0803	0.0958	0.1113	0.0998	0.0736	0.0904
Electricity, hydropower, at pumped storage power plant	0.0053	0.0048	0.0045	0.0022	0.0049	0.0103
Electricity, nuclear, at power plant	0.4562	0.4219	0.3712	0.2898	0.1963	0.1827
Electricity, production mix photovoltaic, at plant	0	0	0	0.0001	0.0003	0.0214
Electricity, at wind power plant	0	0	0.0042	0.0302	0.0726	0.1287
Electricity, at cogen ORC 1400kWth, wood, allocation exergy	0	0.0001	0.0068	0.0163	0.0286	0.0134
Electricity, at cogen with biogas engine, allocation exergy	0	0	0	0	0	0
Electricity, production mix FR	0.0233	0.0365	0.0007	0.026	0.0233	0.057
Electricity, production mix PT	-0.0387	-0.0264	-0.0203	-0.0013	-0.0243	0
TOTAL	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

2.4. Allocation assumptions

The function of the analyzed system was not linked directly to a specific final product, such as harvested grapes or wine bottles. Moreover, it is important to consider that numerous additional products, co-products or residues are formed throughout the winemaking process. However, given the surface-based perspective and the limitation of the system boundaries to the viticulture stage of wine production, the derived environmental impacts were not disaggregated between different material flows. Hence, allocation between co-products was not necessary.

2.5. Life Cycle Impact Assessment

ReCiPe is the assessment method selected for the computation of the environmental impacts. The rationale behind this selection is due to the vast range of environmental impacts that it considers and the possibility of providing a single score weighted endpoint average for the results, which allows tackling priority impact categories based on the values obtained.

3. Results

While the results for this study are still under computation, and will be presented in full at the LCA of Foods 2014 Conference, at this point in the analysis is was possible to anticipate to a certain extent some of the main drivers of environmental impact in the appellation under study. On the one hand, as can be observed in Figure 1, the biogenic GHG emissions for the appellation were computed based on the IPCC guidelines for the entire assessment period. The results show a steady tendency toward a higher potential of carbon storage in the appellation as surface land destined to viticulture decreases. In fact, by year 2008 the carbon storage potential of the land was found to be superior to the biogenic GHG emissions occurring in the area studied.

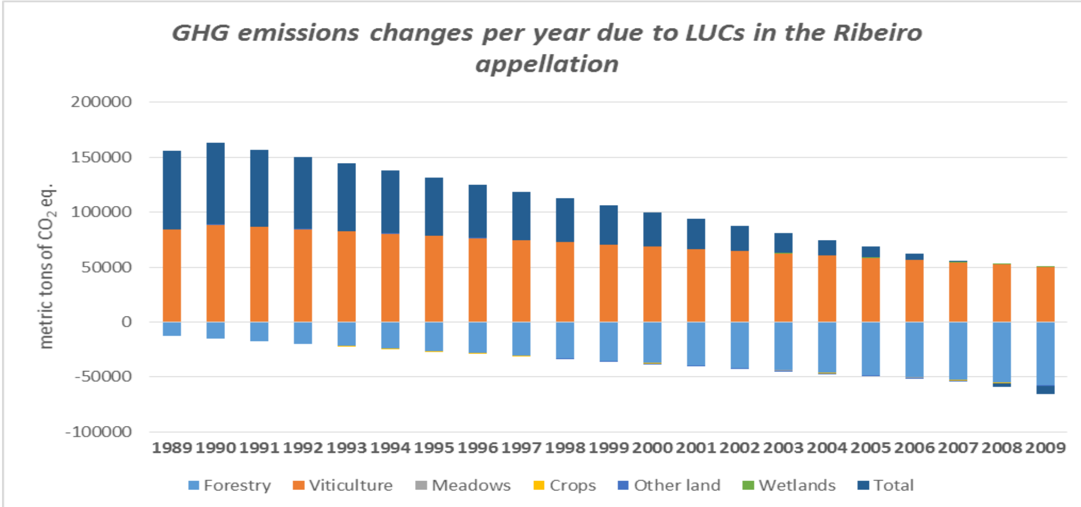

Figure 1. Biogenic greenhouse gas emissions (metric tons of CO2eq) and storage potential of the *Ribeiro* appellation in the period 1989-2010 (Results referred to the FU).

A second issue of interest, was the analysis of the electric mix evolution in Spain in the timeframe analyzed (see Figure 2). As can be seen below, the GHG emissions associated with the production of electricity in Spain ranged from 0.5-0.6 kg CO_2eq per 1 kWh between 1989 and 2005, depending mainly on the proportion of coal used for electricity production, as well as on the hydroelectric power available. However, in 2005, led by the increase in installed power of some renewable energies, namely wind power, but also photovoltaic to a lesser extent, and to an important reduction in the use of coal, the GHG emissions associated with the production of 1 kWh of electricity were down to 0.3-0.45 kg CO_2eq for the period 2009-2011.

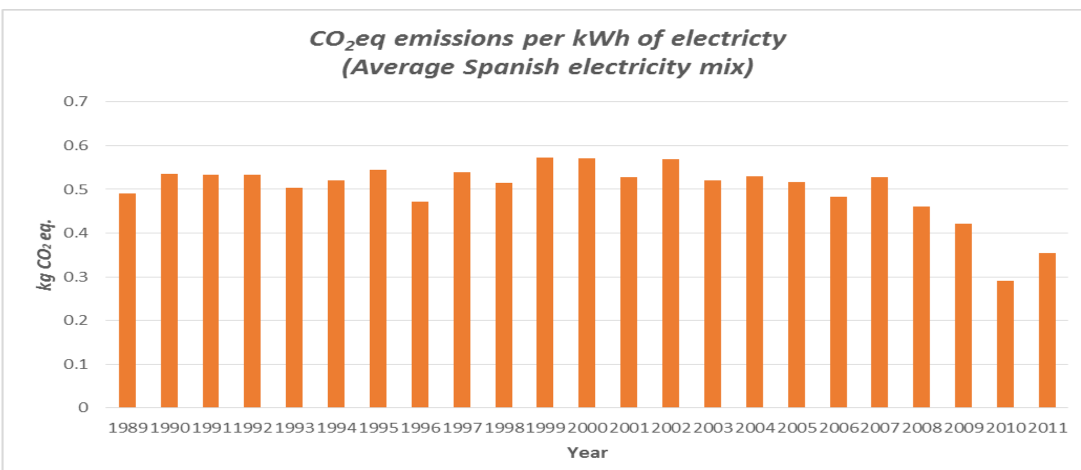

Figure 2. Greenhouse gas (GHG) emissions per kWh produced in the Spanish electricity mix (1989-2011). Data referred to 1 kWh produced, including pumping and international exchanges.

4. Discussion and conclusions

The partial results presented in this paper demonstrate substantial increases in carbon storage in the *Ribeiro* appellation in the past two decades due to LUCs. This is attributable to the lower surface area destined to winemaking, a phenomenon that can be linked to the demographic crisis in the area, but also to a more specialized and technological production system, in which the average producer cultivates more viticulture land than in the late 80s. While these results can seem positive, it is also important to bear in mind that most of the land that has been abandoned in the past 20 years has not received adequate management, but has been absorbed without any human monitoring by the surrounding habitat conditions. Therefore, risks linked to forest fires, as well as a lack of understanding of the optimized carbon storage potential of abandoned land, are two issues that remain essentially unexplored.

In this preliminary analysis of the results, and based on the results gathered in Figure 2, it may appear that the anthropogenic GHG emissions regarding electric production in Spain would also contribute to enhance the trend shown by biogenic emissions. However, this interpretation can be highly biased due to the increase in technological use in viticulture activities. Therefore, while the total GHG emissions per kWh have decrease in this period, especially linked to the higher proportion of renewable energy in the mix (mainly wind power) in recent years, viticulture practices are also using more energy intensive materials in trellis (e.g. concrete or granite slabs, rather than *Acacia dealbata*) or fertilizers (more widespread use of inorganic fertilizers in recent years).

While this results are still partial and preliminary, they do present some promising tendencies for the environmental profile of the appellation. However, numerous issues remain open from an LCI and LCIA perspective. In the first place, the uncertainties in the LCI appear to increase as we go back in the year of assessment, linked to the increasing difficulty to retrieve data. Secondly, the preliminary results advances in this paper have also focused on GHG emissions, ignoring other impact categories used in ReCiPe. While this decision is justified in the fact that both energy and land use are of extreme importance in the assessment of climate change, future updates of this work will have to analyze in depth the trade-offs between impact categories linked to the dynamic changes occurring in the appellation.

Finally, while the results presented already allow certain interpretations from a policy-making approach, the expected final outputs of the study will permit a broad discussion on how different policy decisions can steer the dynamic environmental impact changes in the near future.

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Social sustainability issues of cod and haddock fisheries in the northeast Atlantic

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ABSTRACT

Research on the sustainability of capture fisheries has focused more on environmental and economic sustainability than on social sustainability. To assess social sustainability, relevant and important social sustainability issues (SSIs) need to be identified. The objective of this study was to identify relevant and important SSIs for cod and haddock fisheries in the northeast Atlantic based on two stakeholder surveys. The first survey resulted in the identification of 27 relevant SSIs that were ranked in order of importance in the second survey. Results show that worker safety, product freshness and companies' salary levels are the most important SSIs. Results on the relevance and importance of SSIs enable the industry and policy-makers to direct improvement efforts towards the more important SSIs.

Keywords: social issues, stakeholders, capture fisheries, working conditions, fish welfare

1. Introduction

Sustainability of food production was firmly put on the research agenda with the release in 1987 of the Brundtland report on sustainable development entitled 'Our Common Future' (Brundtland 1987). Sustainable development was defined in this report as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs." This definition captures notions of human needs (economic sustainability), equity concerns (social sustainability) and the carrying capacity of our planet (environmental sustainability).

Research on social sustainability has mainly concentrated on the inclusion of the social dimension in LCA (Benoît-Norris et al. 2011, Dreyer et al. 2006, Hunkeler 2006, Kruse et al. 2009, Norris 2006, Weidema 2006). To this end, indicators have been proposed (e.g. UNEP/SETAC 2010a, b, c, d, e) that address various social sustainability issues (SSIs) (i.e. aspects of social sustainability that are important to consider in an assessment) that are considered to be universal. The importance of SSIs, however, depends on the cultural, political, social and economic context of the case considered (Benoît-Norris et al. 2011, Glaser & Diele 2004).

A social sustainability assessment, therefore, should start with a description of the case considered (Mollenhorst & De Boer 2004, Van Calker et al. 2005). This study concerns a group of cod and haddock fishing companies in the northeast Atlantic. These Norwegian and Icelandic fishing companies employ trawlers, longliners, auto-liners, and Danish seiners in coastal and offshore fisheries to produce fresh and frozen fillets. The second step in social sustainability assessment is the identification of relevant and important social sustainability issues (Mollenhorst & De Boer 2004, Van Calker et al. 2005). Since the importance of SSIs depends on the context of the case considered, stakeholder input should be used (Caffey et al. 2000, Meul et al. 2008, Mollenhorst & De Boer 2004, Van Calker et al. 2005). Stakeholders are those individuals or organizations that can affect or are affected by the activities of the cod and haddock fishing companies in the northeast Atlantic (based on Freeman 1984).

The objective of this study is to identify relevant SSIs for cod and haddock fisheries in the northeast Atlantic and to determine the importance of these SSIs based on stakeholder input. Since social sustainability concerns a diversity of stakeholders with different interests, a heterogeneous group of stakeholders was consulted in order to get a representative set of SSIs (Caffey et al. 2000, Meul et al. 2008, Mollenhorst & De Boer 2004, Van Calker et al. 2005).

2. Methods

Stakeholders were invited to take part in two consecutive surveys on SSIs. These stakeholders were identified from a detailed value chain characterization and distributed across seven distinct stakeholder groups, namely: fishing companies, fishing company employees, suppliers and processors, sales organisations, consumers (represented by consumer organizations), policy-makers, and fish welfare organizations. Each stakeholder group encompasses multiple stakeholders who share similar interests.

The first survey served to compile a long list of relevant SSIs for cod and haddock fisheries in the northeast Atlantic. To this end, respondents were asked to 1) indicate whether SSIs on an initial list of possible SSIs based on literature (Table 1) were relevant, and 2) add new SSIs in case of deficiencies. SSIs on this initial list concern working conditions, terms of employment, employees' job fulfilment, companies' contribution to the local community, food safety and product quality. When at least one respondent indicated that an SSI on the initial list of SSIs was relevant, then this SSI was added to the long list. If any stakeholder mentioned a new SSI, it was added to the long list as long as the SSI was clear, concerned social sustainability and did not overlap with any of the SSIs on the initial list.

Table 1. Initial list of social sustainability issues with the relevant references to literature

SSI	Reference
Employees' working schedule	(Kruse et al. 2009, UNEP/SETAC 2010e)
Arrangements for employees' overtime	(UNEP/SETAC 2010e)
Companies' salary levels	(Benoît-Norris et al. 2011, Caffey et al. 2000, Kruse et al. 2009, Meul et al. 2008, UNEP/SETAC 2010e)
Companies' timely payment of salaries	(UNEP/SETAC 2010e)
Pension fund contributions of companies for their employees	(Kruse et al. 2009, UNEP/SETAC 2010e)
Employees' income security during sickness	(UNEP/SETAC 2010e)
Worker safety	(Caffey et al. 2000, UNEP/SETAC 2010e)
Healthy working environment (ergonomics)	(Mollenhorst & De Boer 2004, UNEP/SETAC 2010e)
Provisions aboard for the crew (for example, sports and internet facilities on vessels that go out to sea for longer periods of time)	(Kruse et al. 2009)
Employees' job satisfaction	(Bavinck & Monnereau 2007)
Employees' professional pride	(Meul et al. 2008)
Freedom of association and collective bargaining	(UNEP/SETAC 2010e)
Equal opportunities for immigrant workers	(UNEP/SETAC 2010e)
Community involvement of cod and haddock fishing companies	(UNEP/SETAC 2010b)
Local ownership of cod and haddock fishing companies	(Caffey et al. 2000)
Local employment from cod and haddock fisheries	(UNEP/SETAC 2010b)
Seasonality of employment	(Kruse et al. 2009)
Product freshness	(Roininen et al. 2006)
External damages to the fish (e.g. cuts)	(based on Mollenhorst & De Boer 2004)
Internal damages in the fish (e.g. bleedings)	(based on Mollenhorst & De Boer 2004)
Microbiological food contamination (from, for example, bacteria and parasites)	(Codex Alimentarius Commission 2009)
Chemical food contamination (from, for example, cleaning materials or pesticides)	(Codex Alimentarius Commission 2009)
Physical food contamination (from, for example, gear or equipment)	(Codex Alimentarius Commission 2009)

The second survey served to determine the importance of each SSI on the long list of relevant SSIs that resulted from the first survey. Respondents were asked to put the five most important SSIs from the long list in order of importance.

The analysis of the second survey focused on responses per stakeholder group in order to correct for unequal numbers of responses between stakeholder groups. For each respondent, five SSIs were scored 1 to 5 points, where 1 point was assigned to the SSI that the respondent ranked lowest and 5 points were assigned to the SSI that the respondent ranked highest. These individual scores were then used to calculate the average score for SSI i and stakeholder group k as follows:

$$\overline{S}_{ik} = \frac{\sum_{l=1}^{l} S_{ilk}}{n_{lk}}, \qquad \text{Eq. 1}$$

where S_{ilk} is the score for SSI i of respondent l in stakeholder group k and n_{lk} is the number of respondents l in stakeholder group k. Then, the overall score for SSI i was calculated as follows:

$$\overline{S}_i = \frac{\sum_{k=1}^{k} \overline{S}_{ik}}{n_k}, \qquad \text{Eq. 2}$$

where $\bar{S}_{ik}$ is the average score for SSI *i* and stakeholder group *k*, and n_k is the total number of stakeholder groups *k*.

3. Results

In total, 41 copies of the first survey were returned from April to August 2013 via mail, e-mail and online. Table 2 reports the number of responses per stakeholder group for the first survey and shows that responses from employees in fishing companies represent the largest share of responses.

In the first survey, each SSI on the initial list of SSIs was selected by at least 5 and at most 33 respondents. This indicated that all SSIs on the initial list had to be included in the long list of SSIs for the second survey. In addition, six new SSIs suggested by respondents were added to the long list, namely: fish welfare during capture, humane slaughter of fish, opportunities for life-long learning, on-the-job training, employees' travel time from home to work and back, and time at home. The new SSI 'fish welfare during capture' implicitly included the former SSIs 'internal and external damages to fish', which were thus excluded from the long list of SSIs. As such, the first survey resulted in a long list of 27 relevant SSIs that was used as input for the second survey.

Table 2. Targeted sample and number of responses for survey 1 and survey 2 per stakeholder group and per stakeholder

Stakeholder group	**Stakeholders**	**Targeted sample (n)[a]**	**Responses survey 1 (n)**	**Responses survey 2 (n)**
Fishing companies	Fishing company owners	7	3	4
	Fishing company associations	1	1	2
Fishing company employees	Fishing companies' employees	81	12	20
	Labor unions	7	1	0
Suppliers and processors	Suppliers to the vessels	0	0	1
	Processing company owners	4	3	4
	Processing companies' employees	335	1	6
	Processing company associations	1	2	2
Sales organizations	Retailers	12	1	1
	Merchants	3	6	1
	Organizations promoting the sector	2	1	0
Consumers	Consumer organizations	1	0	0
Policy-makers	Local municipalities	4	0	1
	Government fisheries department	2	4	1
	Public administration	0	0	1
	Public institutions	0	0	1
	Government unspecified	0	0	1
	Certifiers of stock sustainability	3	1	0
Education	Organizations that provide education to employees in fishing companies	Not applicable	Not applicable	4
Fish welfare organizations	Fish welfare organization	1	1	1
Other interest groups	Other interest groups not specified further	0	4	0

[a] Targeted sample refers to the number of organizations or the number of individuals.

In total, 66 copies of the second survey were returned from October to December 2013 via e-mail and online. Seven surveys were excluded because they were incomplete and no stakeholder affiliation was entered. Eight surveys were excluded because stakeholder affiliation was either unclear or irrelevant. Table 2 reports the number of responses per stakeholder group for the second survey and shows that responses from employees in fishing companies represent the largest share of responses.

The analysis of the second survey focused on the importance of the SSIs on the long list, based on overall scores per SSI. Overall scores for the SSIs ranged from 0 to 1.99, with an average of 0.6. The three most important SSIs are worker safety, product freshness and companies' salary levels, as indicated by the highest overall scores for these SSIs (1.99, 1.46 and 1.34, respectively). The three least important SSIs are seasonality of employment and arrangements for employees' overtime, which were not ranked by any respondent, and equal opportunities for immigrant workers (with an overall score of 0.02).

4. Discussion

The first survey resulted in the identification of 27 relevant SSIs for cod and haddock fisheries in the northeast Atlantic. These 27 SSIs include six SSIs that were not considered before in studies on social sustainability (Bavinck & Monnereau 2007, Benoît-Norris et al. 2011, Caffey et al. 2000, Codex Alimentarius Commission 2009, Kruse et al. 2009, Meul et al. 2008, Mollenhorst & De Boer 2004, Roininen et al. 2006, UNEP/SETAC 2010b, e). The newly identified SSIs address aspects of employees' training and education opportunities, employees' time off from work and fish welfare. The second survey resulted in a ranking of the 27 relevant SSIs in order of importance. This ranking of SSIs shows that the most important SSIs for cod and haddock fisheries in the northeast Atlantic are worker safety, product freshness and companies' salary levels. Worker safety (Kruse et al. 2009, Utne 2007) and companies' salary levels (Kruse et al. 2009) were identified as important SSIs in studies on social sustainability of capture fisheries before, but product freshness was not.

Results on the relevance and importance of SSIs for cod and haddock fisheries in the northeast Atlantic were based on relatively low numbers of responses to the first and the second survey (41 and 51 responses, respectively). Larger numbers of responses to the surveys would have been preferable, but consulting a diversity of stakeholders was prioritized. There were no responses, however, from consumers, one of the seven stakeholders groups distinguished. Therefore, consumer perspectives on social sustainability issues should be addressed in future research.

Responses to the surveys were unevenly distributed across stakeholder groups; the largest number of responses per stakeholder group was 20, whereas the smallest number of responses per stakeholder group was one. Not correcting for this uneven distribution of responses across stakeholder groups would have resulted in a disproportionately large influence of certain stakeholder groups at the expense of other stakeholder groups. Therefore, average scores (indicating relative importance of the SSIs) were first determined per stakeholder group and then averaged over all stakeholder groups.

It is interesting to explore to what extent results on the importance of SSIs for cod and haddock fisheries in the northeast Atlantic are applicable beyond the context of this case. In this study, several different types of fishing techniques were considered (i.e. trawling, seining, auto-lining and long-lining in coastal and offshore waters) within one region. This means that the context of cod and haddock fisheries in the northeast Atlantic is mainly determined by its regional setting. Therefore, similar results on the importance of SSIs can be expected for other capture fisheries (e.g. herring or mackerel fisheries) in the northeast Atlantic and its subareas, and for capture fisheries in similar regions (e.g. the northwest Atlantic).

5. Conclusion

This study resulted in the identification of 27 relevant SSIs for cod and haddock fisheries in the northeast Atlantic. These 27 SSIs include six newly identified SSIs that address aspects of employees' training and education opportunities, employees' time off from work and fish welfare. The most important SSIs are worker safety, product freshness and companies' salary levels. Results presented in this study on the relevance and importance of SSIs for cod and haddock fisheries in the northeast Atlantic inform stakeholders, and especially the industry and policy-makers, about the relevant social sustainability themes and their valuation by different stakeholders. This enables the industry and policy-makers to direct improvement efforts towards the more important SSIs.

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Comparison of process-based models to quantify major nutrient flows and greenhouse gas emissions of milk production

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ABSTRACT

Assessing and improving the sustainability of dairy production systems requires an accurate quantification of greenhouse gas (GHG) emissions and major nutrient (N, C, P) flows associated with milk production at the animal, farm and field-scale. Life cycle inventory databases are, however, often based on rough estimates of GHG emissions and nutrient flows, and cannot account for spatially-explicit variation in these flows. Emission estimates can be improved when underlying processes influencing GHG emissions and nutrient balances are explicitly considered. We aim to improve life cycle inventory databases for milk production in the US by integrating process-based models into LCA data acquisition. We therefore perform a quantitative comparison of five process-based models to determine major nutrient flows and GHG emissions of milk production at the animal, farm and field-scale.

Keywords: milk production, nutrient flows, greenhouse gas (GHG) emissions, process-based models

1. Introduction

The livestock production sector is a key contributor to a range of critical environmental problems, at local, regional and global scales (Steiner et al. 2006, Pelletier and Tyedmers, 2010, Bouwman et al. 2013). Ruminant livestock systems contribute substantially to global warming through greenhouse gas (GHG) emissions. The global dairy sector is presently responsible for 2.7% of total, global greenhouse gas emissions (FAO, 2010). In the US, the agricultural sector is responsible for 8.1% of total GHG emissions (EPA, 2014). After fossil fuel, enteric and manure methane (CH_4) emissions are the second and third most important sources of GHG from the dairy supply chain in the US (Thoma et al. 2013). In addition, crop-livestock production systems are the largest cause of human alteration of the global nitrogen (N) and phosphorus (P) cycles (Bouwman et al. 2013). Total N and P in animal manure exceed global N and P fertilizer use (Bouwman et al. 2013) and livestock manure is a major source of anthropogenic ammonia (NH_3) emission in the U.S. (NRC, 2003) and globally (Steiner et al. 2006).

Assessing and improving the sustainability of dairy production systems thus requires an accurate quantification of greenhouse gas (GHG) emissions and major nutrient (N, C, P) flows associated with milk production at the animal, farm and field-scale. In large, nonhomogeneous countries like the US, milk production practices and climate conditions vary widely, which can result in large, farm-specific variations in GHG and nutrient emissions (Henderson et al. 2013). Life cycle inventory databases are, however, often based on rough estimates of GHG emissions and nutrient flows, and cannot account for spatially-explicit variation in these flows. Emission estimates can be improved when underlying processes influencing GHG emissions and nutrient balances are explicitly considered (e.g. Schils et al. 2004, Li et al. 2012). We aim to improve life cycle inventory databases for milk production by integrating process-based models into LCA data acquisition. We therefore perform a quantitative comparison of five process-based models to determine major nutrient flows and GHG emissions of milk production at the animal, farm and field-scale. We compare these models in terms of GHG and NH_3 emissions to air, as well as nitrate (NO_3^-) and phosphate (PO_4^{3-}) emissions to ground water, and allocate these emissions to the different dairy production components, including five feed crops, animal (enteric) emissions, and manure management. This model comparison study is part of a larger project that aims to reduce the life cycle environmental impact of dairy production systems in the USA (www.sustainabledairy.org). Here, we present the results of the first round of model comparison, which focussed on GHG and NH_3 emissions to air.

2. Methods

2.1. Model description

In the model comparison, we included five process-based models: CNCPS6.1, DAYCENT, ManureDNDC, APEX and IFSM3.4 (Table 1). These models operate on different scales, each having their own unique features. The *Cornell Net Carbohydrate and Protein System (CNCPSv6.1*) model is an animal scale model that predicts changes in N_2O, CH_4, and NH_3 emissions for a wide range of feed, environmental and ration characteristics (Tylutki et al. 2008). The model provides enteric emissions and nutrient balances per cow.

Table 1. Included process-based models

Model	Scale	Reference
CNCPSv.6.1	Animal	www.cncps.cornell.edu/
DAYCENT	Field	www.nrel.colostate.edu/
DNDC-manure	Farm	www.dndc.sr.unh.edu/
APEX	Field to watershed	www.epicapex.tamu.edu/
IFSM	Farm	www.ars.usda.gov/

DAYCENT is a daily-time step, plant-centric soil biogeochemical model (Del Grosso et al. 2001, 2002, 2005). Model outputs include daily fluxes of various N-gas species (e.g., N_2O, NO_x, N_2), daily CO_2 flux from heterotrophic soil respiration, soil organic C and N, net primary productivity (NPP), daily water and nitrate (NO_3) leaching, and other ecosystem parameters. *Manure-DNDC* provides a detailed description of the on-farm biochemical cycle of N and P as well as the use of water for each individual crop (alfalfa, corn, grass, soybean and wheat). The model can be used for predicting crop growth, soil temperature and moisture regimes, soil carbon dynamics, nitrogen leaching, and emissions of trace gases including nitrous oxide (N_2O), nitric oxide (NO), dinitrogen (N_2), ammonia (NH_3), methane (CH_4) and carbon dioxide (CO_2). A specific feature of DNDC is the biogeochemical process model for quantifying greenhouse gas and ammonia emissions from livestock manure systems (Li et al. 2012). *APEX* (Williams et al. 2012) is a comprehensive daily-time step model able to link field to watershed-scale, simulating detailed agricultural management and quantifying productivity as well as impacts on a suite of environmental processes (hydrology, erosion, net ecosystem exchange, soil carbon dynamics, nitrogen balance, etc.) (Gassman et al., 2010). The model can be configured to simulate pertinent management strategies, such as rotational grazing, movement of animals between paddocks, and application of manure removed from livestock feedlots or waste storage ponds. *The Integrated Farm System Model (IFSM)* provides a process level simulation of farm production systems and predicts the performance, economics, and environmental impacts of alternative production practices (Rotz et al., 2012). IFSM provides field scale emissions for individual crops, use of machinery, enteric emission, manure management and other flows from the barn.

2.2 Model comparison – whole farm approach

For model comparison, the whole-farm model approach was used, which is an established powerful methodology to develop GHG mitigation strategies for farming systems (e.g., Schils et al. 2005, Schils et al. 2007, del Prado et al. 2013). The whole-farm approach reveals trade-offs between emissions of the different GHGs and other pollutants, and ensures that interactions between nutrient cycles are taken into account (Schils et al. 2005). All models were run for a dairy farm in New York state, using the same input data. Emissions of NH_3 and important greenhouse gases, e.g. CH_4 and N_2O, were collected for each model. As not all models include the same processes, emissions were allocated to three main farm processes, e.g. barn, manure handling, and field, to facilitate a meaningful comparison of the models. Barn emissions were further segregated into “enteric emissions from livestock” and “other barn emissions”, such as ground emissions. “Field” emissions include all emissions associated with crops and soil, such as soil N_2O emissions.

Total global warming impacts were quantified for each farm process by multiplying the emissions of CH_4 and N_2O with the substance-specific global warming potential (GWP100, IPCC 2007, 1 for CO_2, 25 kg_{CO2eq}/kg_{CH4} and 298 kg_{CO2eq}/kg_{N2O}). These substance-specific global warming impacts can be aggregated to obtain the total global warming impact (in CO_2 equivalents). At this stage, biogenic CO_2 emissions were excluded from the quantification of global warming impacts, assuming that the total biogenic CO_2 input balances the biogenic CO_2 output.

2.3 Pilot farm

Input data collected for the NY state farm, include detailed feed scenarios, use of machinery on the farm, a description of feed crop cultivation practices and a description of the manure management system. A summary of the main farm characteristics is provided in Table 2.

Table 2. Summary of farm characteristics

Dairy cows (number)	**Annual average production (lbs/head)**	**Housing**	**Manure management**	**Crop types**	**On-farm feed production**
1100 lactating 165 dry cows 425 older heifers 500 young heifers	26364	Free stall	Slurry with natural crust	485.6 ha corn 344 ha alfalfa 48.6 ha soybean 80.9 ha small grain 60.7 ha grazing	85%

3. Results

3.1 GHG emissions

First results show that predicted enteric CH_4 emissions dominate GHG impacts at the individual farm level (Fig. 1). Enteric CH_4 emissions, are similar for ManureDNDC v3, IFSM3.4 and CNCPS6.1 and range from $2.0\cdot10^5$ $kgCH_4$/yr to $2.4\cdot10^5$ $kgCH_4$/yr, leading to a dominant GHG contribution between $5.0\cdot10^6$ and $5.9\cdot10^6$ $kgCO_{2equ}$/yr. For the other farm processes, predictions of GHG emissions differ more between models. ManureDNDC and IFSM3.4 differ in their prediction of manure emissions (both CH_4 and N_2O) and barn N_2O emissions. IFSM3.4 predicts a relatively large contribution, i.e. ~20%, of manure CH_4 to total GHG impacts, whereas the contribution of manure CH_4 to total GHG impacts is only 3% in manureDNDC predictions. Alternatively, manureDNDC predicts a relatively large contribution of barn N_2O emissions to total GHG impacts (15%), whereas these emissions are not included in IFSM3.4. Particularly striking is the difference in predictions of field N_2O emissions. The DayCent prediction of $1.5\cdot10^4$ kgN_2O emitted per year is substantially higher than the predictions of ManureDNDC, IFSM3.4, which are $1.2\cdot10^3$ kgN_2O/yr and $1.7\cdot10^3$ kgN_2O/yr, respectively. APEX predicts an emission of $1.9\cdot10^3$ kgN_2O/yr (excluding fallow land), and an emission of $4.2\cdot10^3$ kgN_2O/yr if most of the manure is assumed to be applied on a fallow land. On a per crop basis, other differences between model field N_2O predictions appear (Fig. 2). Both APEX and DAYCENT predict a high contribution of alfalfa to total N_2O emissions, whereas alfalfa contributes only slightly to the total N_2O emissions from the field in manureDNDC predictions. ManureDNDC predicts a dominant contribution of corn (46%) to total N_2O emission. This is consistent with DAYCENT predictions, where corn, next to alfalfa, dominates total N_2O field emissions.

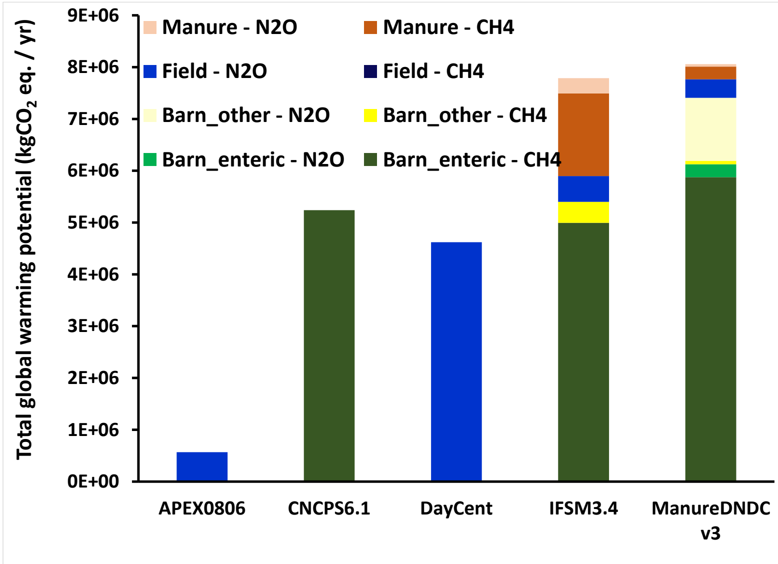

Figure 1. Comparison of total annual global warming impact calculated by five process models for field barn and manure management systems of the pilot NY farm

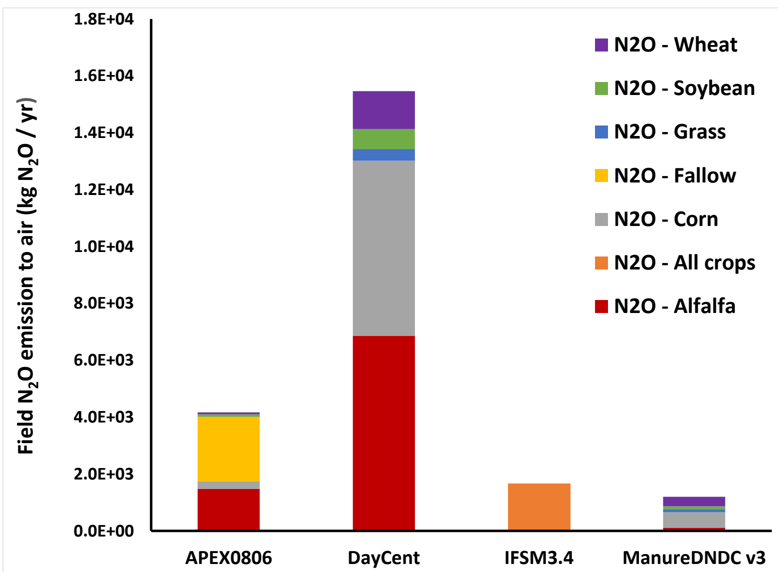

Figure 2. Crop-specific nitrous oxide (N_2O) emission to air for crop production calculated by four process models

3.2 Ammonia emission

This first model comparison show that predictions of ammonia emissions to air are mostly similar across models (Fig. 3). Predictions of barn NH_3 emissions are very close for ManureDNDC and CNCPS6.1, i.e. $4.3 \cdot 10^4$ kgNH_3/yr and $4.5 \cdot 10^4$ kgNH_3/yr, respectively. These emissions are a factor of 1.6 higher than the barn NH_3 emissions predicted by IFSM3.4. Field NH_3 emissions are highly comparable between ManureDNDC, IFSM3.4 and APEX, ranging from $1.4 \cdot 10^4$ kgNH_3/yr to $1.9 \cdot 10^4$ kgNH_3/yr. Manure NH_3 is highly comparable as well. Manure NH_3 emission estimations differ by a factor 1.1 between IFSM and ManureDNDC. CNCPS6.1 predicts only livestock emissions.

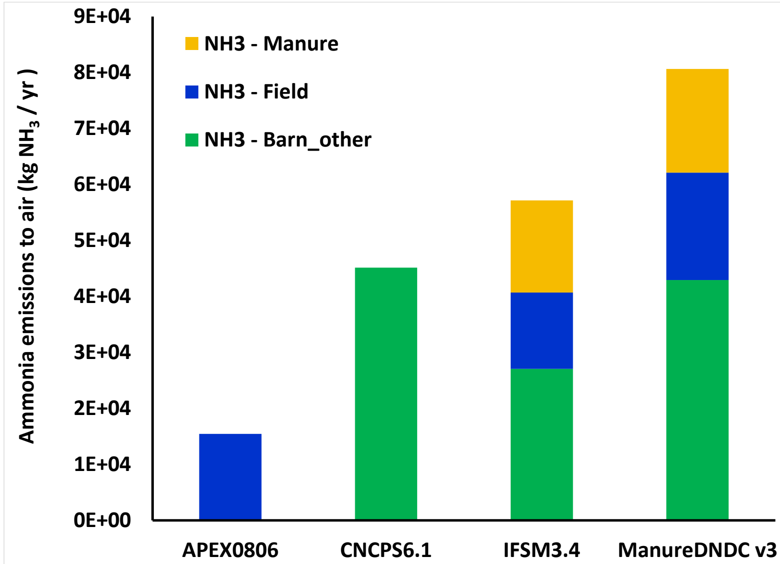

Figure 3. Ammonia emissions to air

4. Discussion

Results of the first round of model comparison show that enteric CH_4 emissions are dominating the total global warming impact at the individual farm level. This finding is consistent with results from other studies. Thoma et al. (2013) showed that enteric CH_4 emission contribute 25% of total C footprint of the dairy supply chain. In addition, DelPrado et al (2013) found that enteric CH_4 and crop land N_2O are the main contributors to whole-farm greenhouse gas impacts for grassland ruminant-based farms system in Europe, although large site- and farm-specific variations were observed.

The different models vary primarily in their prediction of N_2O emissions related to barn and field. One of the reasons for these differences can be that not all models were run with exactly the same input data. A closer inspection of model input data revealed that model input data differ in terms of meteorological data and soil type. Current research efforts focus on further harmonization of input data. A further analysis of model differences will be carried out after the second modelling round with the harmonized input data set. Model validation will then be carried out on another pilot farm for which more detailed experimental data are available. Finally, for improving the process modelling of N_2O flows, the monitoring tasks of the larger project will provide empirical measurements of field barn and manure emissions.

5. Conclusion

A first comparison of five process-based models shows that enteric CH_4 emissions are dominating the total global warming impact at the individual farm level. It is thus necessary to accurately predict and measure these enteric CH_4 emissions. Field CH_4 emissions play only a minor role in the total global warming impact of the individual farm. There may be less urgency to monitor variations in field CH_4 emissions. Model predictions mainly differ in terms of field N_2O emissions, which may partly result from differences in input variables. Current research focusses on further harmonization of the input data for the pilot farm, which will be used for the second model round. In this second round, we will extend the model comparison to nitrate (NO_3^-) and phosphate (PO_4^{3-}) emissions to groundwater. In addition, we will establish and compare whole-farm nutrient (C, N, P) balances. Once models are evaluated and harmonized, integrating process-based models into LCA data acquisition, as proposed here, can provide a more accurate, spatially-explicit approach to derive farm-specific nutrient flows and GHG emissions, which is essential to improve sustainability assessments of (US) dairy production systems.

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Consideration of the product quality in the life cycle assessment: case of a meat product treated by high pressure

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ABSTRACT

New technologies play an important role on sustainable food production and to ensure food security for the next decades (Sonesson 2010). High-pressure processing is a promising emerging technology which was considered environmentally friendly in terms of energy consumption (Toepfl et al. 2006). The technology improves food quality that was not included in previous LCA studies. In this case study, the food safety as a quality characteristic was included in the comparison of the environmental impact between a high-pressure treated ham and the traditional one. At the consumption stage, the food safety was considered as the potential damage to human health by *Listeria monocytogenes*. As an additional step of meat processing, the contribution of high-pressure processing to the life cycle of cooked ham was negligible. However, improving food safety could partly compensate for the environmental impact due to the extended shelf life of meat products.

Keywords: Meat product, high pressure processing, Life Cycle Assessment

1. Introduction

Sustainable food processing is a critical element for the future generations to ensure food safety and to preserve food quality with efficient use of resources (Floros et al. 2010). The impact of high pressure processing on food production has been recently explored (Davis et al. 2009; Pardo and Zufía 2011). Previous studies have focused on high pressure processing without considering new characteristics of high-pressure treated products.

Commercialized high-pressure treated products are mainly vegetable products (33%), meat products (30%), seafood and fish (15%) and beverages (12%) (Mújica-Paz et al. 2011). Within those food categories, meat products were reported to have the greatest environmental impact (Tukker et al. 2006). Moreover, there is little knowledge about the environmental impact of high pressure processing on meat products.

Nutritional quality has been considered in sustainable approach of life cycle assessment as functional unit (Shau and Fet 2008). However, other quality features should not be neglected in efforts to evaluate the environmental impact of products. Food safety is a food attribute which is associated to health risk (Röhr et al. 2005). Novel preservation technologies improve food safety and potentially decrease the human risk associated to foodborne illness (Hugas et al. 2002). Likewise, the food recall for microbiological criteria is an important wastage that could be avoided by emerging technologies.

Listeria monocytogenes is a pathogen of public health concern, responsible of listeriosis. For the characteristics of production and consumption, ready-to-eat foods are considered to be a vehicle of listeriosis foodborne (Zhang et al. 2012). High-pressure processing is an alternative to control *Listeria monocytogenes* without using growth inhibitors and post packaging pasteurization. For this reason, we develop an approach to consider the food safety in the comparison of the high pressure treated product with the traditional one. The objective of the study was to evaluate the environmental impact of a sliced ham treated by high pressure and to compare it with the untreated product, considering the quality of products.

2. Methods

2.1. Goal definition, system description and data collection

The life cycle assessment was performed to compare the environmental impact between the sliced cooked ham treated by high pressure and the conventional product. The functional unit is 1 kg of sliced cooked ham consumed. The economic value of the traditional product and the high-pressure treated one was also used to compare their environmental impact. The prices of the products were recorded in countries where the technology is currently employed on commercialized sliced cooked ham.

From a cradle-to-crave perspective, the system consists of 4 subsystems: meat production, cooked ham production, distribution and consumption. Figure 1 shows the studied system and the high-pressure processing as an additional step of cooked ham production. The data were collected from a meat producer (slaughtering and cooked ham production) in France and a high-pressure equipment manufacturer. The data for the meat production in France were obtained from Agribalyse database 1.1. Capital goods, cleaning products and packaging material for transportation were not included. Mass allocation was performed for multi-products system. The high-pressure treatment of the packaged sliced cooked ham is 600 MPa for 3 minutes at room temperature in order to eliminate post processing contamination of *L. monocytogenes*.

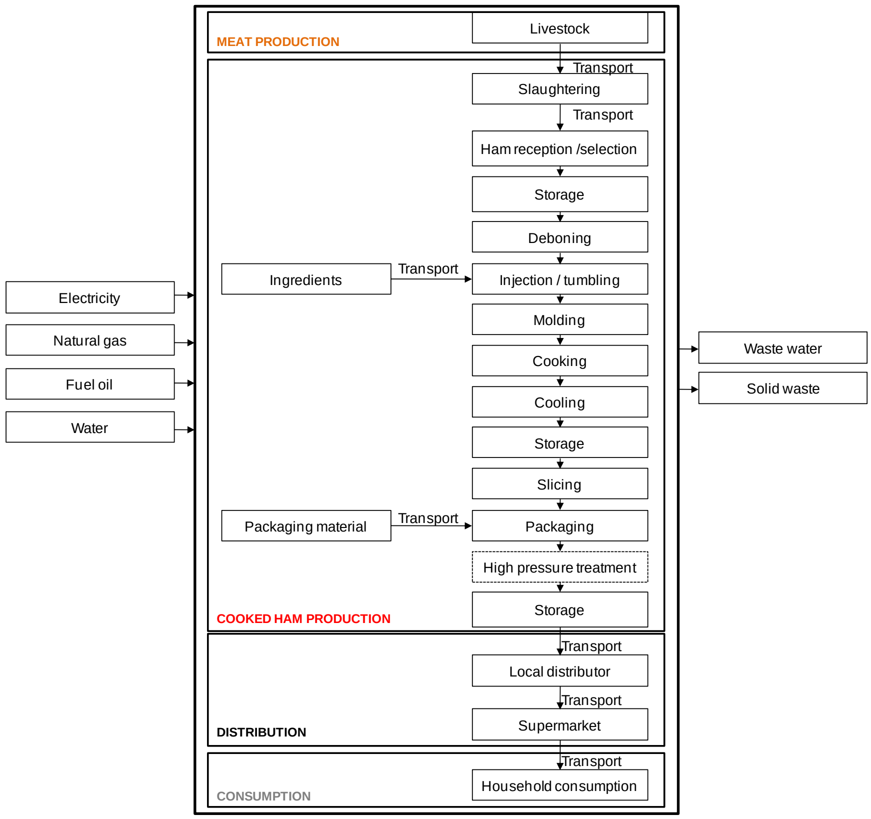

Figure 1. Flow diagram of the life cycle of sliced cooked ham

2.2. Food safety: Health effects due to the exposure to *L. monocytogenes*

Food safety is a food attribute, presented as damage to human health. Food quality in the ready-to-eat meat product is evaluated in terms of the disease burden caused by *L. monocytogenes*. The model refers to the change of Disability Adjusted Life Years (DALY) upon exposure to the pathogen. The comparison between the meat products provides the potential benefit of using high-pressure processing as a listericidal treatment. We proposed to consider this aspect on the human health damage category.

The impact of using high-pressure processing to control *L. monocytogenes* was evaluated by using the risk assessment of *L. monocytogenes* in ready-to-eat food (WHO/FAO, 2004) and the BRAFO tiered approach (Hoekstra et al. 2012). The framework consists of three parts which are: the exposure assessment, the dose-response and the risk characterization.

The exposition factor is the quantitative evaluation of the ingestion probability of *L. monocytogenes*. The prevalence and contamination level of *L. monocytogenes* on ready-to-eat meat products from the report of the European Food Safety Authority (2013) were assumed. The distribution of *L. monocytogenes* in the ready-to-eat meat product was obtained from the serving size (g) and the distribution of the contamination level (European Food Safety Authority 2013).The serving size (g) was calculated from the annual consumption of cooked ham in France. The serving size was assumed to be the same to women and men. The dose is estimated by using the Equation 1.

$$\text{Dose(cfu)} = \text{Average serving(g)} \times \text{Contamination level(cfu/g)} \qquad \text{Eq. 1}$$

The probability of illness ($P_{illness}$) at a specific dose (16 g/day) is obtained from the distribution of *L. monocytogenes* on the ready-to-eat meat (Dose) and the probability of *L. monocytogenes* of causing an infection to a susceptible population (r: 5.85 x 10^{-12}) (WHO/FAO 2004), as presented by the exponential dose-response relation (Eq. 2).

$$P_{\text{illness}} = 1 - e^{-rDose} \qquad \text{Eq. 2}$$

The probability of listeriosis due to the annual consumption of packaged sliced ham in France at the prevalence and contamination level reported is 5.1 x 10^{-8}. The probability to develop a specific manifestation was estimated by using equation 3. The factor (F) is the proportion of cases of listeriosis which develop the specific manifestation and it was obtained from government publications and literature data (Institut de veille sanitaire 2012, Lecuit and Leclercq 2013).

$$P_{meningitis/\,septicemia/\,neo-natal listeriosis} = P_{\text{illness}} \times F_{meningitis/\,septicemia/\,neo-natal listeriosis} \qquad \text{Eq. 3}$$

Listeria monocytogenes is responsible for two forms of listeriosis: non-invasive and invasive listeriosis. The invasive listeriosis is the most severe manifestation form and it may induce meningitis, septicaemia and neonatal infection. In this study, the invasive listeriosis was only considered. The severity of listeriosis was estimated in terms of the indicator of the human health impact category (Disability-adjusted life years, DALY). Similar approach was used to evaluate the impact of foodborne disease on public health in Greece and Netherlands (Gkogka et al. 2011, Havelaar et al. 2012).

Listeriosis, in the invasive form, was evaluated in the susceptible population (older adults and pregnant women). The DALY estimation was according to "BRAFO tiered approach" (Hoekstra et al. 2012) for quantal health effects. The expression (Eq. 4) considers the health recovering, the decease and the sequelae of individuals, which was calculated for each manifestation of listeriosis (meningitis, septicemia and neo-natal listeriosis).

$$\text{DALY}_{a,s} = P_{effect}\,(P_{rec} \cdot \text{YLD}_{rec} \cdot w + P_{die}(\text{YLD}_{die} \cdot w + \text{LE}_{a,s} - \text{CA} - \text{YLD}_{die}) + (1 - P_{die} - P_{rec}).(\text{LE}_{a,s} - \text{CA}).\,w) \qquad \text{Eq.4}$$

The DALY $_{a,s}$ is the disability-adjusted life years for a specific age (a) and sex (s). The P_{effect} is the probability of illness by listeriosis. The P_{rec} is the probability of health recovering (Aouaj et al. 2002, Dzupova et al. 2013, Goulet et al. 2012). The P_{die} is the probability to die due to the illness (Varon 2009, Goulet et al. 2012). The YLD_{die} and the YLD_{rec} is the illness duration for the individuals who die and recovers health, respectively (Gerner et al. 2005, Fernandez et al. 2011, Kemmeren et al. 2006). The w is disability weight of the disease (Melse et al. 1998, Melse et al. 2000). The CA is the actual age of diseased individual. The LE is the life expectancy for the individual with an age CA. The Table 1 summaries the parameters used for each manifestation. For neo-natal listeriosis, parameters values ($P_{rec,}$ YLD_{rec}, YLD_{die}, P_{die} and *w*) were assumed since it represents the neonatal death and the abortion due to this manifestation. The data for the concerned population in France were obtained from the INSEE (National Institute of Statistics and Economic Studies). The annual DALY is estimated and summed for each individual of the susceptible population.

Table 1. Values of parameters used for DALY calculation

Diseases	**F**	**P_{rec}**	**YLD_{rec}**	**P_{die}**	**YLD_{die}**	***w***
Meningitis	0,28	0,565	0,500	0,22	0,080	0,310
Septicemia	0,65	0,791	0,020	0,209	0,080	0,930
Neo-natal listeriosis	0,315	0	0	1	0	1

The baseline scenario is the current characteristics of distribution and prevalence of *L. monocytogenes* on ready-to-eat meat products (European Food Safety Authority 2013). The worst scenario is based on a challenge test in a cooked ham at the limit of pathogen inactivation by high pressure processing (Myers et al. 2013). It was assumed a lower prevalence of contaminated products (0.001%) for the worst scenario and the potential human health damage due to listeriosis was evaluated for 14 days of storage time at the retail store.

2.3. Life cycle impact assessment

The life cycle assessment was conducted by using the SimaPro 8.0.2 software. Two methodologies of impact evaluation were employed: problem-oriented method (CML IA v 3.0 and Cumulative energy demand v 1.08) and damage-oriented method (Eco-indicator 99 v 2.09). The environmental impact categories selected in the analysis were non-renewable energy, global warming potential, acidification, eutrophication, and photochemical oxidation. The impact categories for the human health damage were also included.

2.4. Sensitivity analysis

A sensitivity analysis was performed to evaluate the impact of input variables on the human health damage due to listeriosis. Likewise, the impact of capital goods and the extended shelf life of products were considered.

3. Results

3.1. Environmental impact of meat products

Figure 2 shows the contribution of each stage of the life cycle of the meat product untreated (traditional product) and treated by high pressure processing (HP product). The results of the stage contribution to environmental impact categories are expressed in mass of products (vertical axis). The environmental impact of products is quite similar in this case, while no changes of post-processing stage were described. Meat production is the main contributor for the life cycle of the meat products. For the traditional product, 45.3% of non-renewable energy use, 73.1% of global warming potential, 92.4% of acidification, 87.7% of eutrophication and 89.4% of photochemical oxidation were due to meat production. It is in accordance to the composition of final products which contains about 90% of pork meat. Besides, cooked ham production was responsible for 27.5% of the non-renewable energy use and 15.6% of the global warming potential. The contribution of cooked ham production was 5.7%, 9.9% and 6.4% for acidification, eutrophication and photochemical oxidation, respectively. High pressure processing as an additional operation to meat processing represents an increase of 0.7% of non-renewable energy demand. The contribution of high pressure processing was equal or less than 0.1% increase on the results for global warming potential, acidification, eutrophication and photochemical oxidation. The distribution and the consumption phase accounted for 15% and 12% to non-renewable energy use. For global warming potential, the contribution of distribution and consumption was 9% and 2% while for acidification, eutrophication and photochemical oxidation contributed equal or less than 2%.

Food quality is related to human perception and how the product fulfills their expected requirements (Röhr et al. 2005). Considering food quality implies to include its multiple aspects. The results are also presented in terms of the economic value since it reveals the emotional value of a product (Shau and Fet 2008). The economic value of products (price of packaged sliced cooked ham in Europe) is used to compare the environmental impact between the traditional and the treated product as presented in Figure 2. The environmental burden of different impact categories decreased about 16% on the treated cooked ham in comparison to the traditional one.

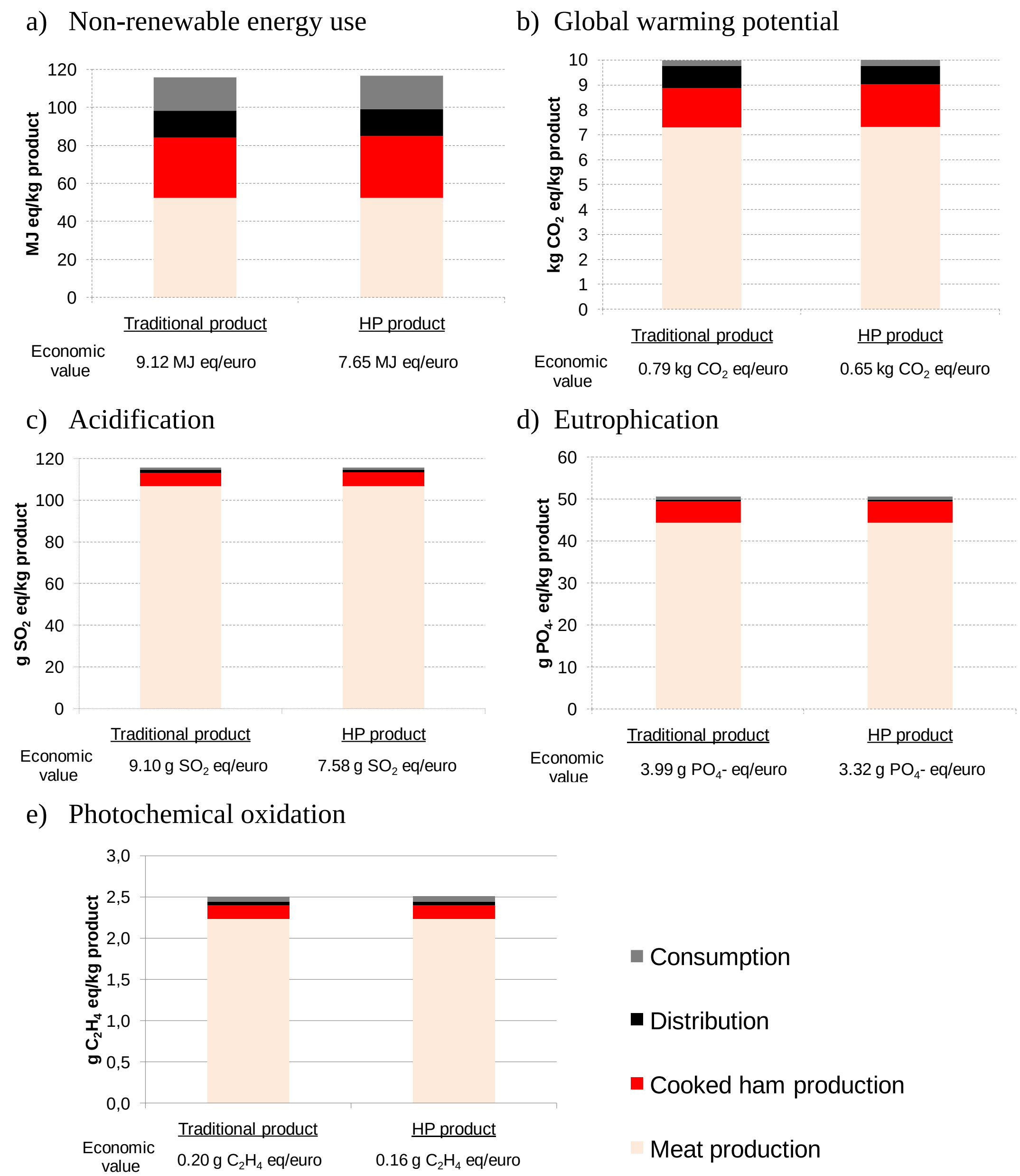

Figure 2. Contribution of stages to the environmental impact categories for the life cycle of the sliced cooked ham per functional unit: bar graphs present results per kg of product and below the figure, the results are presented per euro spent for product.

3.2. Food safety: Health effects due to the exposure to *L. monocytogenes*

The total annual health impact of the exposure to *L. monocytogenes* by the meat product for the susceptible population in France is 2.84 DALY [2.24 – 3.63]. The risk of listeriosis among the older adult population contributed about 66% to the total impact.

The table 2 reports the results of impact categories for human health per 1 kg of product. The baseline considered the actual impact of listeriosis on the susceptible population in France. High pressure processing is able to inactive *L. monoctogenes* up to 10^4 cfu/g (Bajovic et al. 2012, Myers et al. 2013). Considering the actual characteristics of prevalence and contamination of *L. monocytogenes* on the ready-to-eat meat products, high pressure is an efficient listericidal treatment if it is used as final step of meat processing.

In relation to the damage impact categories, the actual exposition to the pathogen presents a more relevant impact for human health than the consequences associated to ozone layer depletion for the life cycle of the meat product. Human health damage is mainly caused by carcinogens, climate change, respiration inorganics and radiation. Minor changes were observed in the endpoint impact categories of the treated product.

For the worst scenario, the potential damage due to listeriosis increases with the pathogen growth at storage conditions. It becomes more relevant than health risk due to exposition to organic particles after 12 days. No growth of *L. monocytogenes* in treated products reduces the potential human health damage due to sliced cooked ham.

Table 2. Results of impact categories of human health damage category for the life cycle of the sliced cooked ham per functional unit

Impact category	**per 1 kg of product**	**Baseline**		**Worst scenario - traditional product**				**HP product**
		Traditional product	**HP product**	**0**	**7 days**	**12 days**	**14 days**	**14 days**
Carcinogens	DALY	2.69E-06	2.69E-06	2.35E-06	2.46E-06	2.51E-06	2.53E-06	2.54E-06
Respiration-organics	DALY	7.55E-09	7.55E-09	7.05E-09	7.27E-09	7.29E-09	7.30E-09	7.30E-09
Respiration-inorganics	DALY	8.87E-06	8.87E-06	8.51E-06	8.71E-06	8.76E-06	8.78E-06	8.78E-06
Climate change	DALY	2.03E-06	2.03E-06	1.81E-06	1.98E-06	1.99E-06	1.99E-06	2.00E-06
Radiation	DALY	1.21E-07	1.23E-07	6.62E-08	8.94E-08	1.05E-07	1.11E-07	1.13E-07
Ozone layer	DALY	6.17E-10	6.19E-10	5.75E-10	5.97E-10	6.00E-10	6.01E-10	6.03E-10
Listeriosis	**DALY**	**2.05E-09**	**0.00E+00**	**5.04E-11**	**1.73E-10**	**9.80E-09**	**4.48E-08**	**0.00E+00**

3.3. Sensitivity analysis

The sensibility analysis showed the influence of product contamination, consumer behavior and characteristics of the concerned population on the results of human health damage due to the microbiological risk; which were evaluated individually. The variations of the prevalence of *L. monocytogenes* on meat products from the actual level (2.07%) increased the human health damage results; about 48.6% for increasing 1% the prevalence level of contamination. The changes on the consumer consumption also influence the human health damage results and it was proportional to the magnitude of serving size change; about 6.3 % for increasing 1 g the actual serving size.

Considering the projection of population growth from government publications, to 2050, the population in France would increase 6.7% and the older adult (60 years and more) population would represent 31.9% of the estimated population. The human health damage results for the projected population increased 36.6% the indicator value. The increase of older adult population was the main contributor of the indicator variation.

On the other hand, the differences of the environmental load between the traditional product and the treated product increase less than 1% if capital goods are included in the analysis.

The extended shelf life of the treated meat product can increase the storage time at the supermarket. If we consider two-fold the baseline of the storage time, the use of non-renewable energy increases 10% and less than 1% for other impact categories in comparison to the traditional product.

4. Discussion

4.1. Contribution analysis

The meat production has the greatest contribution to the life cycle of meat products. Cooked ham production is the second contributor to environmental impact. The contribution of this stage refers to energy consumption and refrigerant leakages. High pressure processing has a little influence on the environmental profile of the sliced cooked ham if there are no modifications of the behavior at the distribution and consumption stages. Main differences between meat products concern to the storage time that can be prolonged in treated products. The treated meat products with extended shelf life increase the use of non-renewable energy without important change of global warming potential due to electricity mix in France.

4.2. Food quality in the life cycle assessment

The treated meats products commercialized are value-added products (Bajovic et al. 2012). The average price of treated sliced cooked ham and untreated ones recorded in Europe was considered in the analysis. The difference between prices was observed in data collection. Consumers with knowledge of high pressure processing are able to pay an additional cost for the features of the high-pressure treated product (Hick et al. 2009). High pressure processing provides added value to meat products by improving food safety since it allows not only the inactivation of pathogenic microorganisms, but also the reduction of use or the removal of food preservatives.

The comparison of the value performance with regards to the environmental impact is defined as eco-efficiency (Schau and Fet 2008). For treated products, each monetary unit as benefit represents less environmental load. In consequence, the use of high pressure processing on cooked ham production improves the eco-efficiency of the meat product. In fact, the comparison between the traditional and the treated product is greatly influenced by the choice of functional unit.

Food safety is also presented as the human health impact of the disease associated to *L. monocytogenes*. Including other dimensions of sustainability at the endpoint level could lead the identification of the main concerns to human health. Although uncertainties of endpoint methods are large, the comparison of impacts at the endpoint level shows that carcinogens, climate change, effects of respiration inorganics, radiation are the most important contributors to damage to human health. However, the human health effect due to listeriosis is relevant in comparison to effects to human health by ozone layer depletion.

For the worst scenario, the human health effects due to listeriosis raised importantly in two weeks; followed by the radiation impact category which is associated to radiation exposition by the power production of nuclear plants. High pressure processing can prevent the years of life lost due to the disease at the baseline (2.84 DALY) and for the worst scenario (14 days: 62.22 DALY). It is important to note that the results of human health effects of listeriosis correspond to specific geographic conditions, consumer characteristics and for a susceptible population (older adults and pregnant women). The benefit of preservation technologies are not only limited to the actual state of food safety of meat products; but also in the prevention of higher human health damage due to food-borne diseases. Efforts are necessary to decrease the principal contributors to the environmental impact and therefore, the damage to the human health. Reduction of meat content in meat products is complicated without affecting its nutritional value. Finally, novel technologies play a role in minimizing losses by improving food quality.

5. Conclusion

Novel technologies have a great responsibility to improve sustainability on food processing. In this case, increasing food processing steps to improve food quality does not necessary cause a notable increase of environmental load of processed food. The eco-efficiency of the meat product can be enhanced by using high pressure processing. In fact, the food quality can be improved without important contribution to the environmental impact of the meat product. Furthermore, the direct impact of food safety on one of protection areas of LCA, human health, was described. The benefits of high pressure processing on food safety could compensate in part the use of non-renewable resources due to the extended shelf life of products. Other quality features should be included in life cycle assessment for improving comparison between products.

6. Acknowledgements

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Co-digestion of dairy cow manure and food waste creates a more efficient biogas cycle

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ABSTRACT

The AgSTAR project of the U.S. EPA analyzed the possibility of installing anaerobic digesters with energy generation and nutrient capture in all confined dairy operations in the US with 500+ cows, with a total potential capacity of more than 2,000 digesters nationwide. This study uses environmental life cycle assessment to identify the potential benefits of disposing locally accessible commercial food waste with manure at those digesters (via co-fermentation) in comparison to other disposal options. Anaerobic digesters show an advantage compared to compost or landfill for all impact indicators examined: human health, climate change, ecosystem quality and water withdrawal. For greenhouse gases, the installation of digesters could potentially reduce 20-25 million metric tons (MMT) CO_2e of fugitive methane emissions from manure compared to current manure management practices, 10 MMT CO_2e of fugitive methane emissions from landfilled food waste, and another 10 MMT of CO_2e from avoided electricity, fertilizer production and peat moss production by harvesting energy, nutrients, and fibers from digesters, respectively.

Keywords: Dairy, food waste, anaerobic digestion, compost, landfill, manure

1. Introduction

It is estimated that 13 percent of greenhouse gases in the United States are associated with growing, manufacturing, transporting, and disposing of food (US EPA 2009). This sector is therefore among the high priorities for achieving climate reduction goals. As part of the next phase of the 2013 Climate Action Plan, announced in March 2014, the US government has placed a heavy focus on cutting methane emissions to help meet a national goal of reducing U.S. greenhouse gas emissions by 17 percent below 2005 levels by 2020. Two key sources of methane reductions cited in the strategy are the reduction of landfill waste and a plan to manage biogases. The latter component mentions implementation of manure digesters on dairy farms as one key component.

Economic analysis has previously shown a large potential benefit for implementation of digesters on US dairy farms (Informa 2013), emphasizing in particular the potential economic benefits that might be gained through taking advantage of the excess capacity of these digesters to produce fuel from other organic matter, such as collected food waste. In addition, the financial potential to market the co-product of digesters as a landscaping material (replacement to peat moss) is also explored. In total, the potential economic benefits of implementing these systems on dairy farms are estimated to be in the range of $3 billion.

Complementary to the economic analysis, the objective of this research was to use environmental life cycle assessment to validate and to examine the effectiveness of the strategy of diverting food waste to co-digestion with dairy manure as a solution for reducing greenhouse gas emissions and other environmental impacts. The system analyzed forms an example of how a circular economy concept is possible within the food production chain, with wastes from food production re-entering the food production system in a way that supports the solution environmental problems at earlier stages of production. The system also provides linkages to other industrial systems, resulting in a potential for a type of Eco-industrial Park centered around an agricultural system. The quantification of the benefits of this system is based on a life cycle assessment (LCA) approach).

2. Methods

2.1. Scenarios under study

Worldwide, the dominant methods of waste disposal are landfills and open dumps. According to U.S. EPA 2012 MSW Characterization Reports (US EPA 2012a), food waste is the largest category of municipal solid waste (MSW) sent to landfills in the United States, accounting for approximately 21% of the waste stream. In

2012, over 30 million tons of food waste is sent to landfill in the US. Of the less than 5% of food waste currently being diverted from landfills, most of it is being composted to produce fertilizer. In this study, the following three systems are analyzed:

a. Landfilling (baseline)
b. Composting
c. Co-digestion of food waste with dairy manure.

In addition to these scenarios for the management of food waste, the benefit is also assessed of using the fiber derived from digesters as a replacement for peat moss in landscaping applications. The present scope does not include consideration of the impact in producing any of the treatment infrastructures for the various options.

2.2. Landfilling

Figure 1 illustrates the methane emissions from landfills based on data provided by US EPA (2012 b). Approximately, 38% of methane is directly emitted to atmosphere, 33% of methane is either flared or oxidized, and 29% of methane is recovered for energy, of which most (74%) is harnessed for electricity generation, 24% is used for heating, and 2% is used for alternative transport fuel (US EPA 2012c) LMOP database).

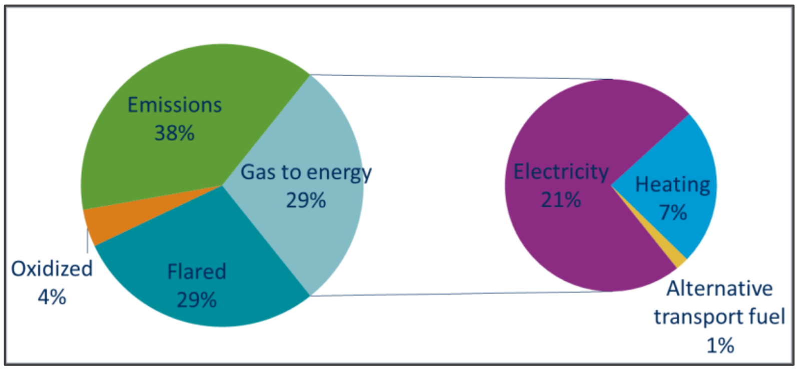

Figure 1. Fate of CH_4 generated by U.S. landfills in 2009

The environmental impact of landfilling food waste was modeled based on an LCA approach using the ecoinvent v2.2 database (processes "disposal, biowaste, 60% water to municipal landfill/CH U") with adaptations made to account for 84% degradation of the materials within a 100 year timespan, following the estimates made in the US EPA (2006) solid waste report. The direct methane emission factor is calculated to be 0.085 kg methane per kg of wet food waste mix treated. The avoided electricity generation, heating and transport fuel were also modeling with the ecoinvent 2.2 database with adaptation from US LCI database and GREET Model.

2.3. Composting

There are several types of composting technologies, including Backyard or Onsite Composting, Aerated (Turned) Windrow Composting, Aerated Static Pile Composting and In-Vessel Composting. ISWM-TINOS (2010) and Morris et al (2011) review recent LCA publications related to composting for more than 30 different sample sizes, results are found with large variability, especially for energy requirement and fugitive emissions. For energy requirement, Rob van Haaren, et al (2010) reported an estimate of 30kWh/ton of wet food waste mix, which is in line with the average energy requirements reported by Diaz et al. (1986, cited in US EPA 2000 report) that estimates 34.4 kWh is required per ton of waste in an MSW composting facility. For fugitive CH_4 and N_2O emissions, default factors from biological treatment for Tier 1 method given by IPCC (2006) were used that is 4 g CH_4/kg waste treated and 0.3 g N_2O/kg waste treated on a wet weight basis. In this study, life cycle inventory data is based on the Rob van Haaren, et al (2010) study for windrow composting technology with fugitive CH_4 and N_2O emission adjusted with data mentioned above.

2.4. Anaerobic digestion

The AgSTAR project (US EPA 2011) analyzed the possibility of installing anaerobic digesters in confined dairy operations of 500 cows or larger, which, at the time of their analysis, amounted to 2,647 dairy operations nationwide. The anaerobic digestion scenario here considers the primary electricity output and co-product (N, P, fiber) produced by disposing of food waste in dairy farm digesters that are also managing manure. The characterization of this system made here relies heavily on the Informa (2013) report National Market Value of Anaerobic Digester Products. This representation attempts to reflect a realistic implementation of the use of digesters for disposal of all accessible food waste within the U.S. on an annual basis. It is assumed that the food waste used in digesters will primarily be commercial food waste and food processing wastes, rather than residential food waste; also the food waste generated on farm (pre-harvest food waste) is not included. Table 1 below lists the potential products for a manure and food waste co-digestion system. Table 2 gives CH_4 emission factors of different feedstock inputs used for anaerobic digesters.

Table 1.Potential Production and Value of Products and Co-Products for co-digestion of manure and food waste in 2,647 Dairy Anaerobic Digesters.

Inputs and assumptions[a]	**Manure+ Food waste**	**Manure**	**Food waste**	**Unit**	**Avoided burden**
Number of cows	3,974,143	3,974,143		Number	
Manure	108,792,165	108,792,165		US tons/year	
Food waste	19,849,474		19,849,474	US tons/year	
Output					
Electricity production	11,701,222	6,375,637	5,325,585	MWh/year	Electricity production, US average
Recovered Nitrogen	331,163	176,337	154,826	US tons/year	Nitrogen production, US average
Recovered Phosphorus	108,782	57,174	51,608	US tons/year	Phosphorus production, US average
Nutrient enriched Fiber	30,111,422	30,111,422		US tons/year	Horticultural peat moss

[a] Data comes from Informa (2013) report.

Table 2.Methane emission factors for different composition of inputs for anaerobic digesters

Type of input	**Quantity**	**Unit**
Manure only (wet)	0.0101	kg CH_4/kg treated
Manure and food waste (wet)	0.0140	kg CH_4/kg treated
Food waste mix only (wet)	0.0353	kg CH_4/kg treated

The methane calculation follows the calculations done in the Ag STAR study (US EPA 2011). The calculation is based on volatile solids (VS) excretion by cattle. This calculation includes a weighted average methane conversion factor based on the manure management systems used on these dairies. EPA provided those estimates by state, but they effectively reduce the methane used in the calculation of the GHG credits by a factor ranging from 35% to 76%, with a national average of 58%.

2.5. Modelling impact of operating digesters and avoided burdens

2.5.1. Impact of operating digester systems and avoided burden from energy and nutrient products

The impact of operating digester systems are modelled based on the ecoinvent v2.2 dataset “electricity, at cogen with biogas engine, allocation exergy/kWh/CH” with adaptations to reflect the biogas generation from food waste.

Methane emissions from dairy manure are assumed to be 100% emitted to air if they are not managed by anaerobic digesters. The avoided burdens associated with energy and nutrient production are modeled with ecoinvent v2.2 datasets as described in Table 1.

2.5.2. Avoided burden from replacing peat moss with digester-derived fibers

There are three main uses for anaerobic digested fiber from dairies. Digested fiber is used for animal bedding, land application to provide plant nutrients, and as a peat moss replacement (Informa 2013). Here, we assume all digester fibers produced are used as peat moss replacement. The Informa report made state-by-state assumptions regarding the fiber that would be available as a peat moss replacement, estimating that of the 18,843,757 cubic yards of fiber produced by plug flow digesters, 16 million cubic yards could be used as peat moss replacement.

Table 3 provides the volume consumed for different types of horticultural peat moss consumption in US in 2011 based on USGS statistics. In total, nearly 6.8 million cubic yards of peat moss is consumed annually in US for horticulture. Most of this volume is imported from Canada. The fiber that goes to peat moss replacement will be 52.8% of the total digester fiber produced. In this study, the maximum benefits are estimated based on future scenario assuming the peat moss market will reach 16 million cubic yards.

Table 3. Different type of peat moss consumed in US in 2011 (USGS statistics)

Type of peat moss consumed in US 2011	**Quantity[a] (cubic yards)**
Canada Sphagnum moss	6,503,569
US Sphagnum moss	289,057
Other import	190,881
Peat moss total	6,792,626

[a] Source: USGS statistics http://minerals.usgs.gov/minerals/pubs/commodity/peat/

The peat life cycle includes harvest, packaging, transport, use, disposal, as well as the in-situ decomposition from the harvest site. For environmental impact of horticultural peat moss production and use, we compared both European (Quantis 2012a) and Canadian peat moss (CIRAIG 2006) studies. Both studies show that peat decomposition during use and disposal stages contributes the majority of climate change impact across the peat moss life cycle (175-180 kg CO_2e/ m^3 of black peat moss). Contributions from other life cycle stage are relatively small or negligible in the European peat moss study. However, in-situ decomposition contributes ~60 kg CO_2e/m^3 peat moss in Canadian horticultural sphagnum peat study, which accounts for peat oxidation from the opening of the harvesting site until its restoration, considering that 50% of the sites are restored after the product cycle and 50% are rehabilitated. Detailed study on peat moss replacement can be found in Quantis (2012b) report (URL: http://www.usdairy.com/~/media/usd/public/digesterfiberstudyquantis.pdf.pdf)

2.6. Impact assessment method

The impact assessment methodology used in this study is based on IMPACT 2002+ vQ2.21, adapted by Quantis to included Water Withdrawal indicator that is the sum fresh water (drinking water, irrigation and sea water. Our results will focus on climate change, water withdrawal and three areas of protection (human health, ecosystem quality and resource depletion). Although the presentation of results focusses on these endpoints, a complete interpretation should also consider the contributing results at the midpoint level and some discussion of midpoint results is included here where relevant to the interpretation. Further details are found at http://www.quantis-intl.com/pdf/IMPACT2002_UserGuide _for_vQ2.21.pdf

3. Results and Discussions

3.1. Detailed results for climate change

Figure summarizes the estimated climate change benefit (CO_2e) of co-digestion of food waste and dairy manures that could potentially be provided by 2,647 digesters, if they were installed in large US dairy operations nationwide and produced the products used in the ways described above. Each bar shows the contribution of a different aspect of the digester + avoided burdens system to the total net impact. Beneficial contributions appear on the positive y-axis, while detrimental contributions appear on the negative y-axis. Compared to landfill of food waste and direct methane emissions from manure, the potential benefit of co-digestion of 18 Million MT of food waste with manures in 2647 dairy farm digesters in US for a year equates to avoiding release of greenhouse gases equivalent to 50 million metric tons of CO_2e.

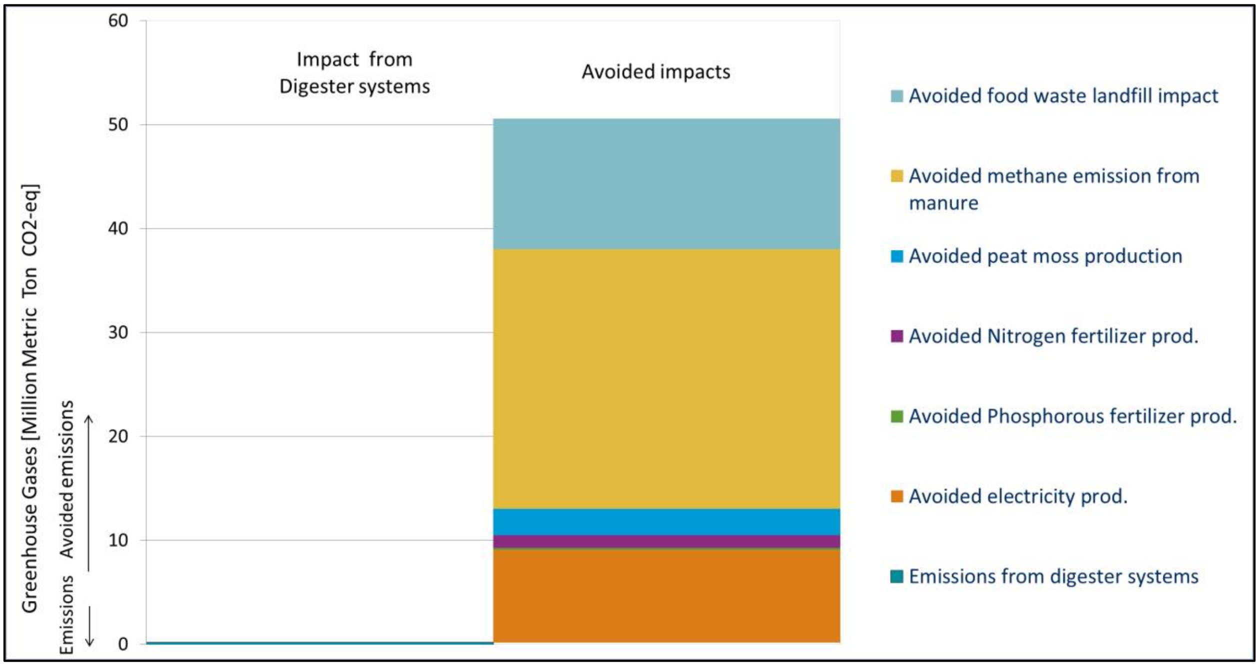

Figure 2. Contribution of different life cycle stages to climate change benefits for co-digestion of food waste with manures system vs. food waste landfill & direct methane emissions from manure (Positive value: beneficial/credits; Negative value: detrimental/impact)

For all scenarios, the avoided fugitive methane emissions from dairy manure management provide the largest climate change benefit (25 million metric ton CO_2e), by converting all biogenic methane that would otherwise be emitted into air to energy. Diversion of food waste from landfill provides the second largest climate change benefit (12 million metric ton CO_2e). US EPA (2012b) reports that 38% of landfill methane is emitted directly to air, and avoiding this release also provides substantial climate change benefit. The third largest climate change benefit (9 million metric ton CO_2e) is due to using the biogas harvested from digesters to replace electricity production.

The impacts of producing the biogas and then converting it to electricity are shown in Figure 2 as 'Emissions from digester systems'. It should be noted that landfill gas is harvested from landfills in the US at a rate of roughly 28%, and largely converted to electricity. When food waste is diverted from landfills, the beneficial use of the landfill gas is also lost. This loss of recovered landfill gas to energy has been subtracted from the food waste landfill benefits shown.

Finally, during the digestion process, nitrogen and phosphorous are recovered, and as a result the digester systems are credited with the avoided impact of average nitrogen and phosphorous fertilizer production in the US. Table Table 4 details the potential positive and negative (beneficial and detrimental) environmental impacts of food waste and manure disposal in landfills, compost and digesters, and the difference among these scenarios. Landfill and composting of food waste generates environmental impact for climate change and ecosystem quality but provides a net benefit for resources due to the harvested landfill gas for electricity and heat, which avoids the fossil fuel use associated with conventional sources for electricity and heat. Composting also shows a net impact in the category of human health effects of pollution, whereas landfilling shows a small net befit in this category. In contrast to landfilling and composting, digestion of food waste and manure causes benefits for all the impact categories examined here.

Table 4: Potential environmental profile of baseline landfill and compost relative to dairy farm co-digestion system (Positive value: beneficial/credits; Negative value: detrimental/impact)

Impact category	Unit[b]	Baseline: Landfill	Alternative: Compost	Alternative: Digester	Compost vs. landfill	Digester vs. Compost	Digester vs. Landfill
Climate Change[a]	MMT CO_2 e	-13	-3.4	37	9.1	41	50
Human Health	DALY	0.22	-4.3	7.2	-4.5	11	7
Ecosystem	Million PDF.m^2.yr	-140	-700	1400	560	2100	1500
Resources	Million MJ Primary	11	1.7	190	-9.7	190	180
Water Withdrawal	Million m^3	17	7.1	340	-10	330	320

a.Climate change result for digester systems also includes 21 MMT CO_2e. benefits from avoided methane emission from dairy manure management. By excluding this benefit, digester is still preferable to other options
b. Abbreviations: MMT= million metric tons

The results indicate that digester disposal is preferable to landfill and compost for all environmental impact categories when biogas is used for electricity generation.

3.2. Summary of results for endpoint impact categories

Figure 3 provides impact results for indicators in addition to climate change for all options under evaluation. The comparative advantage of anaerobic digesters also holds for all endpoint environmental impact categories examined.

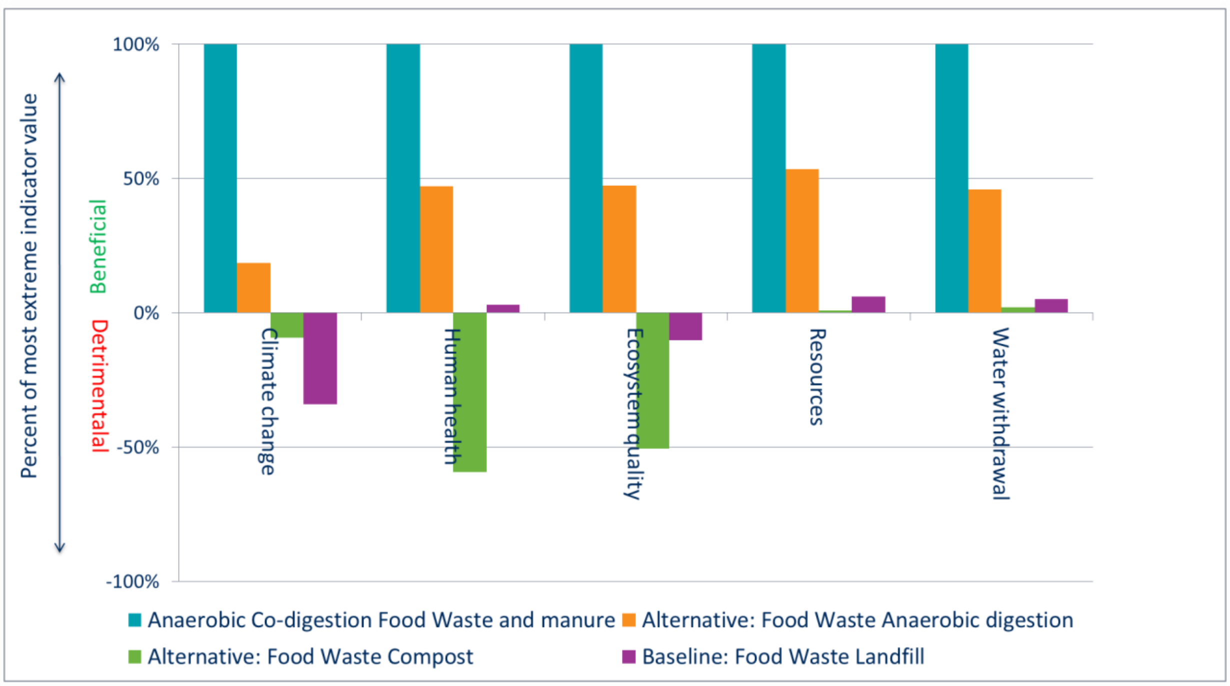

Figure 3. Multi-indicator environmental evaluation of routes of disposing food waste (and/or manure) (IMPACT 2002+ VQ2.2)

4. Conclusion

The findings indicate that for each impact category examined (including climate change, human health, ecosystems and resources), co-digestion of food waste with dairy manure is preferable to landfill disposal and composting for most environmental indicators. In terms of climate impacts particularly, diversion of this food waste from landfill to digesters could avoid the release of greenhouse gases equivalent to between 50 million metric tons of CO_2e. This equates to taking 10 million cars off of the road for a year or to reducing U.S. annual methane emissions by 7%.

Dairy farm digesters are proven to be not only a financially cost-effective but also environmentally favorable business model to provide opportunities for dairies, food processors and retailers. The digesters can play an important role to cultivate the circular industrial economy for food and agriculture industry and also an effective strategy to support achieving greenhouses gas reduction goals in this U.S.

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A food system approach for the identification of opportunities to increase resource use efficiency

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ABSTRACT

The UNEP International Resource Panel has the objective to evaluate the current and projected use of natural resources and as well as to identify opportunities for improvements, using a food systems approach. These opportunities will probably not only include more technical and process oriented opportunities as typically identified with a LCA-approach, but will also include opportunities as the consumption side as reducing food losses and dietary changes. The concept of 'food system' includes all processes and infrastructure involved in feeding a population in a certain region; the global food system is composed of many, partly connected, local and regional food systems. The notion that food systems actors as food processing companies and retailers not only influence the how and where food is being produced, but also people's eating habits and diets, is gaining momentum. Other than in the food chain concept, which perceives the different actors in a more neutral way, the food system concept acknowledges that the actors influence each other, within a certain political, technological, environmental, cultural and institutional setting. Food systems have environmental, socioeconomic and health outcomes, which can, for example in case of undesired outcomes, lead to changes in the context. Given the large differences in regional food systems, a number of global regions (as sub-Saharan Africa, South-East Asia and Europe) are studied in more detail, partly by means of expert workshops.

Keywords: food system, sustainability, resource efficiency

1. Introduction

To most people in the world, current food systems deliver ample and safe food on a day-to-day basis, which can be regarded as a great achievement. This is possible thanks to a large number of actors as farmers, fishermen, processors, retailers and restaurant, as well as to the distribution sector. These actors together make a the food system in a certain region: the concept includes all processes and infrastructure involved in feeding a population in a certain region. The global food system is composed of many, partly connected, local and regional food systems. Governments support the functioning of food systems, for example by regulation on food safety and by investments in the knowledge infrastructure. However, still more than 842 million people are undernourished (FAO 2013), while the same time many people suffer from obesity and other food-related diseases mainly due to unhealthy eating habits. At the same time, food systems are a major user of natural resources, such as land, water and minerals (nutrients) (Foley et al. 2005; Molden 2007), as well as a major source of emissions (as greenhouse gases, pesticides and nutrients) (Bouwman et al. 2009; FAO 2006; Vermeulen et al. 2012). As a consequence of this, food systems are the main driver of global loss of biodiversity (PBL 2010). Due to growing population, increased welfare and urbanization, the environmental impacts are expected to increase. Due to the same drivers, vast changes in food systems are expected, such as supermarketization in developing countries (Reardon et al. 2012). Supermarketization not only affects the supply chain, but very often eating habits and product sourcing as well.

Life cycle assessments (LCA) are widely used to improve processes, support policy decisions and provide a sound basis for informed decisions. LCA can point at technical options to improve production processes and can provide objective and balanced information to compare different production techniques (for example organic versus conventional farming) and different products (for example beef versus poultry meat). Over the last several decades the scope of LCAs has been extended from more traditional subjects such as the use of energy and minerals to more complicated issues including greenhouse gas (GHG) emissions (de Vries and de Boer 2010), land use (Koellner et al. 2013; Mila i Canals et al. 2007); and even social issues as human well-being (Weidema 2006). LCAs of food products are usually quite complex for a number of reasons, but not limited to,:

(i) food production has a large number of environmental effects, including energy, land and water use, use and losses of plant nutrients (N, P and about 15 others), pesticides, GHG; (ii) the generation of co- products and by-products by agri-food production processes; (iii) the site specificity of agricultural production; (iv) the difficulty of defining the functional unit, especially when comparing different types of food, as for example calories or proteins do not cover all nutritional aspects and (v) the various social, cultural and economic aspects of agriculture and food. Thus, in spite of the many efforts to develop LCA for food systems (see e.g. Cowell 1998; Milà i Canals 2003; Brandão 2012), LCAs have been criticized for not being inclusive or complete enough, or for having a too "hard" structure biophysical focus not capable of capturing the wider socio-economic context or "soft" power relationships in value chains (Garnett 2013; Sim 2006). LCAs provide a system for systematically quantifying the relative costs of production for different products and production systems, and for analyzing the contributing factors.

To overcome a number (but certainly not all) of the issues the concept of 'food systems' might be helpful both in identifying opportunities to increase (overall) resource efficiency of food systems as well as to identify food system actors who could facilitate in taking advantage of these opportunities. This food system approach should not be seen as an alternative to LCAs, but as a complementary approach. Especially for more detailed analysis, LCAs remain very helpful, while the food system approach can place the results in a broader context.

The notion that 'food systems' not only influence how and where food is being produced, but also people's eating habits and diets, is gaining momentum in the literature (Ericksen et al., 2010; Ingram, 2011; Pinstrup-Andersen and Watson II, 2011; Vermeulen et al., 2012). In contrast to the food chain concept, which perceives different actors in a more neutral way, the food system concept acknowledges that the actors influence each other, within a certain political, technological, environmental, cultural and institutional context. The objective of the UNEP International Resource Panel (IRP) is to evaluate the current and projected use of natural resources in regional food systems and the global food system as a whole. It also intends to identify opportunities for resource efficiency improvements using a food systems approach, for example by pointing at specific opportunities for certain food system actors. As part of this initiative, the IRP is collaborating with LCA experts in order to gain insights from this well-established methodological approach and cross-fertilize each discipline.

2. Methods

As the concept of 'food systems' and their interaction with natural resources are important, first a conceptual framework was developed, to help further structuring the research questions (Figure 1). Main elements of the food system, and their interactions with natural resources, and environmental and societal impact are depicted in a conceptual framework. Food systems usually can be divided into various food system activities, ranging from provision of inputs (as fertilizers and machinery), primary production (mainly by farmers and fishermen), food trading and processing (crushing of oil seeds, sugar refinery), food industry (preparation of 'food' as eaten by consumers, from bread to ready meals), retailing and food service, consumption and finally processing of food wastes. The different steps can often not be clearly separated, and also very much depending both on the types of food as well as on the regional food system. Food chains in rural areas in developing countries are often relatively short, especially in the case of subsistence farming, where most of the food production and processing (as milling and baking) is within households. In developed countries food systems are typically much more complex. Food systems necessarily depend on certain natural resources, such as land, water and minerals in the case of agriculture, or fish stocks in the case of fisheries. Due to emissions and the use (and sometimes overuse) of resources, food systems have environmental impacts. Life cycle assessment typically addresses the natural resource use and environmental impact of one type of product from input up to the point where it is sold to the consumer and/or consumed. Food systems also include aspects as food system outcomes (effect of food security and human health, farmers' income etc.) which affect general social welfare. Finally, and maybe most importantly, food systems research postulates that the food system are shaped by food system actors (as farmers, food companies and retailers). These actors operate in a certain socio-economic context. The food system is therefore not a neutral logistical food chain as the food system actors have large interests, which basically shape food systems. From a societal point of view, the food system outcomes are most important: to which extent do food systems deliver food security, incomes and are they capable of doing so not only now, but in the future as well.

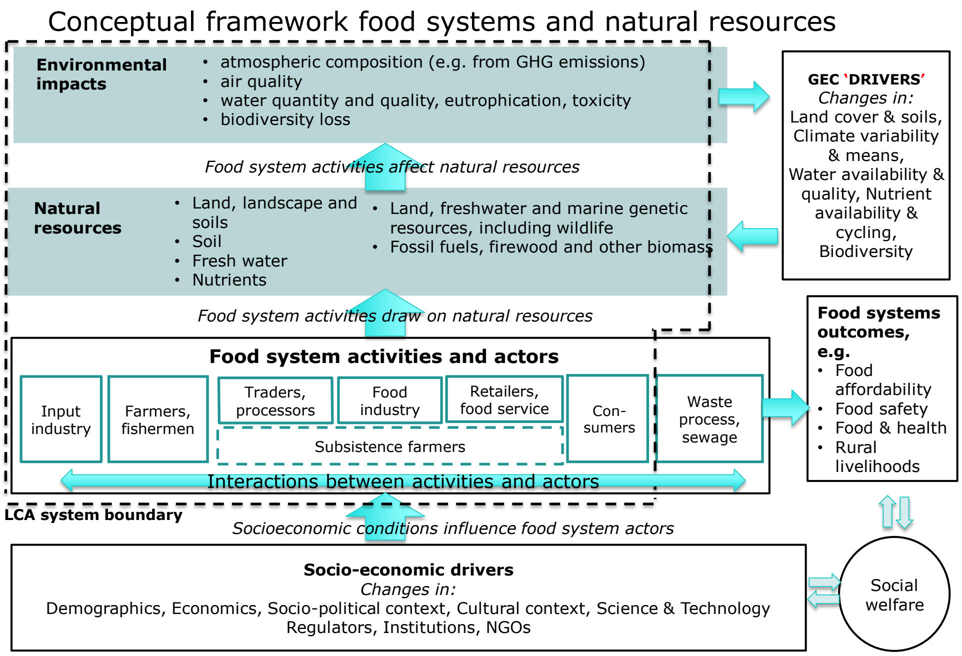

Figure 1. Conceptual framework of food systems, their interaction with natural resources and environmental impacts and food systems outcomes. Also the system boundaries of a typical food LCA study are indicated.

In order to be able to fulfill the objective, for main research question were defined: (i) What is the present and projected use of resources in the various regional food system; (ii) what are the options in biophysical terms to make this reduce this use, or to make the use more sustainable; (iii) how are current food systems functioning, in terms of institutions, technology and relationships (iv) what are the major opportunities within food systems for the various actors to improve the resource efficiency of food systems. The study was mainly based on existing literature, added by expert knowledge. Part of the input of expert knowledge was organized in the form of regional workshops.

Ad (i): The use of resources and environmental impacts were assessed for both the present as well for the future situation. For this agro-economic projection, such as the medium term FAO-OECD Outlook (OECD and FAO 2013) as well as scenario studies focusing on natural resource use have been used (Bouwman et al. 2009; Schmitz et al. 2014). The focus was on a limited number of resources: land, minerals, water, genetic resources and marine resources and environmental impacts (GHG, emission of minerals, biodiversity). Ad (ii): Also based on literature, the major options have been identified to improve the resource use efficiency or to reduce the environmental impacts. These not only include supply-side options (mainly related to production), but also demand-side options as the reduction of food wastes or dietary shifts. Ad (iii): Given the large differences in regional food systems, a number of global regions (as sub-Saharan Africa, South-East Asia and Europe) were being studied in more detail, partly by means of expert workshops. Ad (iv). Based on the previous collected information, the major opportunities for food system actors were identified. The opportunities were not only evaluated in terms of impact on resources use (for which amongst others an LCA-type information was used), but also in terms of broader societal outcomes, where aspects such as food security, health, resilience of farmers and food sovereignty were assessed.

3. Results

Results as synthesized from peer-reviewed articles and state-of-the-arts reports will be used to analyze the present and projected use of resources. Where possible, a disaggregation per region or per commodity will be made, in order have a better understanding of the drivers of resource use and environmental impacts. Examples of the latter are available for nitrogen use and losses (Leip et al. 2013) and greenhouse gas emissions (Lesschen et al. 2011).

In a second step the potential of biophysical options will be evaluated. Where possible and logic the options will be differentiated per region, as not all options are relevant for each region. Both options and the demand side as well on the supply side will be evaluated. The effect of the various options on the selected natural resources and impact will be assessed, when possible based on peer-reviewed articles or otherwise expert judgment. Examples of demand side options are the reduction of food losses (FAO, 2013; Rutten et al., 2013) and dietary shifts (Stehfest et al. 2013; Stehfest et al. 2009; Westhoek et al. 2014). A first evaluation shows that options at the demand side typically have a positive effect on the use of all resources and environmental impacts (Table 1). Supply side options might have certain trade-offs on other resources. An example is increasing crop yields by increasing fertilizer input, which could lead to higher nutrient losses if this is not done correctly (Mosier et al. 2004; Yang 2006).

Table 1. Provisional estimated effect of a number of biophysical options to increase the resource efficiency of food systems, or to reduce the environmental impacts.

	Reduction of resource use			Reduction of environmental impact			
	Land	Water	Minerals	GHG emissions	Water pollution	Bio-diversity	Examples
Demand side							
Reduce food losses and waste	+	+	+	+	+	+	Reduce post-harvest losses
Reduce consumption of livestock products	+	+	+	+	+	+	Reduce portion size, hybrid products
Supply side							
Increase crop yields	++	+?	+ / -	+	-?	+	Improved fertilization; precision farming techniques; improved seeds
Sustainable land management	++	+	+	+	+	+	Soil and water conservation practices
Improve recycling minerals, including reduction of emissions	0?	0?	++	+	++	++	Improved integration of animal manure in crop production
Increase feed efficiency livestock	++	++	++	++	+?	++	Better feeding techniques
Improve transport efficiency	0	0	0	+	0	0	Smart logistic

Consequently, the current food systems in different regions will be examined, as well as projected changes in these. Reardon and Timmer (2012) distinguish food systems in traditional versus modern, while pointing at an intermediate system too. Key findings for Sub-Saharan Africa suggest the trends of urbanization and "supermarketisation" becoming prevalent in the region, which will lead to a significant change of the more traditional food systems. Among others, these trends are leading to changes in diets, increased food wastage at the retail stage, and an increase of globally sourced food in detriment to local sources. The vast majority of energy used for cooking depends directly on local biomass, which is linked to deforestation; in addition 60-70% of this energy is lost in the cooking phase due to inefficient appliances, pointing to a significant potential improvement opportunity.

In terms of opportunities, the food systems approach highlights opportunities which might result from better cooperation between food system actors, leading for example to reduction of food wastes and losses and to a better recycling of nutrients. Other opportunities might arise when downstream actors help farmers to adopt better practices, which might lead to higher crop yields and higher farmers' income. One example is when smallholder farms would be better connected to urban markets, which could not only lead to higher incomes, but could also reduce food losses (through better storage and logistics) and facilitate the provision of inputs as fertilizers. The workshop also identified the role that large companies and global supply chains may play through investing in sustainable sourcing certification schemes. Capacity building to mainstream Good Agricultural Practices linked to these certification schemes is an essential element to enhance yields and reduce environmental impacts related to excessive nutrient losses and/or inadequate pesticide use. In addition, certification facilitates access to markets, which is usually found as a key reason behind pre- or post-harvest food losses. A last example of an opportunity of improved resource use within food system would be the case where retail and food service companies help consumers by making the healthier and environmentally logical choice for consumers. Apparently, corporate interests are not always aligned with optimal societal outcomes of food system. This might be addressed by governments if it is better known under which conditions private actors food systems can deliver better outcomes.

4. Discussion

The food systems approach facilitates consideration of the perspectives of all different actors in the food value chain, and thus the identification of improvement opportunities leading to enhanced resource efficiency, social fairness and economic benefits, in the delivery of the key outcomes. By identifying the key actors of change in specific regional systems, the most adequate opportunities may be leveraged. When considering these actions, the interests of the different key actors should be understood. These interests are often the key to understanding the way the current food system operates, and why certain societal undesired outcomes occur.

Whereas LCA already has clearly defined methods and procedures, the methodology to assess the effect of food systems and potential changes in these, mainly still has to be developed. Given the complexity of food systems, one can even doubt whether a full evaluation can ever be possible. Nevertheless, some aspects or components of food systems can be modeled, for example by using bio-physical or macro-economic models, or combination of these. Examples are the attempts to model on food prices and food security interventions as changes in trade regimes (Anderson 2010; Schmitz et al. 2012; Verburg et al. 2009), investments in agro-food systems in developing countries (IAASTD 2008) and biofuel policies (Bouët et al. 2010; Hertel et al. 2013). Some other studies have focused more on the biophysical aspects, like studies on potential changes in dietary patterns, looking especially at reducing meat consumption (Stehfest et al. 2013; Stehfest et al. 2009). Besides quantitative methods, a large number of more qualitative methods exist, based in thinking from disciplines as political science, sociology and other behavioral sciences and institutional economics. The food system approach has the potential to expand the identification of opportunities for improving the sustainable use of natural resources beyond the more food product based approach as is typically done with LCA studies. Examples of such opportunities in a more bio-physical sense are the reduction of food losses, dietary shifts towards less meat. Moreover, the food system approach offers new perspectives to address a number of issues by involving other agents of change such as food companies and retailers. This could lead to the identification by scientists of opportunities for both policy makers as for actors in the private sector policy-making, thus enhancing the science-policy interface, and maximizing the potential impact of improvement opportunities. Whereas LCA-type analyses have a well-defined methodology, the methodology for food systems analysis largely still has to be developed. This means that the two approaches should be seen as complementary.

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Livestock meat processing: inventory data and methods for handling co-production for major livestock species and meat products

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ABSTRACT
This study analyzed impacts on global warming, energy demand and consumptive water use from meat processing of four major species, chicken, pork, sheep and beef. We investigated four different approaches for handling co-production between meat and non-meat co-products; allocation based on either i) biophysical / protein utilization (BIO); ii) economic value (ECON); system expansion (SE), and a hybrid approach utilizing both biophysical allocation and system expansion (BIO-SE). The impact from processing varied depending on species and impact category, but for all species, the choice of method used to handle co-products had a substantial impact on results. Impacts attributed to the meat product were lowest when BIO was applied, while the highest impacts were generally associated with ECON. SE results were not uniform across the species or across impact categories. Carcase and human edible yield is also discussed as an important consideration when comparing meat products at intermediate stages in the production supply chain.

Keywords: meat processing, life cycle assessment, meat chicken, pork, sheep meat, beef

1. Introduction

Numerous LCA studies have been conducted for meat production systems in the past 10 years, and many of these (e.g. Williams et al. 2006, Cederberg et al. 2009) have focused on the primary production system to the cradle-to-farm gate section of the supply chain. Meat processing, while being a relatively smaller contributor to impacts than the farming section, is a very important stage in the supply chain from the perspective of handling co-production. Livestock systems generate various important co-products, including leather, pet food, protein meals and tallow. Mass loss during meat processing also results in an increase in emissions attributable to each product irrespective of the method used for handling co-production.

A number of farm-gate livestock LCA studies (Cederberg et al. 2009; Leinonen et al. 2012; Opio et al. 2013; Williams et al. 2006) have reported results using a carcase weight (CW) or even a boneless meat functional unit, creating a mismatch between the functional unit and system boundary. This approach induces two errors; firstly it fails to include impacts associated with the meat processing stage of the supply chain and associated transport, and secondly the method attributes all impacts to the meat product, ignoring the value of the co-products entirely. This is inconsistent with other stages in the livestock system where considerable attention has been given to addressing co-production, such as approaches for handling milk and meat in dairy cattle systems (i.e. Cederberg & Stadig 2003, Flysjo et al. 2012). Processing inventories for LCA purpose have been included for chicken meat (e.g.Bengtsson and Seddon 2013), pork (Nguyen et al. 2011), sheep meat and beef (e.g.Peters et al. 2010; Ridoutt et al. 2012). However, few details were provided in these studies and co-production was often allocated on economic or mass bases. In addition, system expansion of meat processing has not been investigated in detail, and thus the significance of rendering was not fully understood. In a recent study of environmental impacts of rendering, Ramirez et al. (2011) considered animal by-products as wastes and do not include the environmental burden of their production. This ignored the fact that meat processing produces not only meat products for human consumption but also pet food and other valuable products. To the authors' knowledge, the significance of co-product handling has not been thoroughly investigated before.

In this paper, inventory data for meat processing of four species (chicken, pig, cattle, sheep) were collected from meat processing plants in Australia covering greenhouse gas (GHG) emissions, fossil energy demand and consumptive fresh water use. The impact of four methods for handling co-production (biophysical allocation based on utilized protein and energy, economical allocation, system expansion and a hybrid approach utilizing both biophysical allocation and system expansion) were presented at the point of meat processing. The results were compared with allocating all impacts to the meat product only. We suggest here a re-definition of the functional unit for meat products to include the yield of edible co-products that are an important contribution to human food supply.

2. Methods

2.1. Goal and scope

The goal of this study was to provide factors and inventory data representative of modern meat processing facilities in Australia, and to explore the impacts of co-production at the point of meat processing. The focus of the study is meat processing, but to illustrate the relevance of methods for handling co-production, the system boundary includes all stages up to the point where product is ready for transport to wholesale or retail (the meat processing plant loading dock). The functional unit is 1 kilogram of retail portions excluding packaging. The study was based on primary data collected from meat processing plants from a series of studies conducted by the authors (Wiedemann et al. 2010; Wiedemann et al. 2012). These studies also provide details of the cradle to farm-gate stage of the supply chain which have been updated to include direct land use change (dLUC) impacts in the present analysis.

The study determined GHG emissions using the IPCC AR4 global warming potentials (GWP) of 25 for methane and 298 for nitrous oxide (IPCC 2007). Energy demand was assessed using the fossil fuel energy demand indicator (Frischknecht et al. 2007) and measured in mega joules (MJ) using Lower Heating Values (LHV). Consumptive fresh water use refers to evaporative uses or uses that incorporate water into a product that is not subsequently released back into the same river catchment. In the meat processing, consumptive water includes raw water drawn from river/bores/reticulation and used in the processing plant, and subsequently treated via waste water treatment. The water subsequently used for irrigation (without going through sewage system) is not considered consumptive usage. Consumptive water use was assessed following (Bayart et al. 2010). Inventory data are reported for impacts relating to GHG emissions, energy and water while co-product handling methods were compared for GHG and energy demand only.

2.2. Processing inventory data

Inventory data were collected from a minimum of three meat processing plants for each species (Table 1, Table 2). Data from the case study processing plants were validated using a recent survey of resource use from beef and sheep meat processing plants (GHD 2011) for important inputs such as electricity, gas and water to improve the representativeness of the dataset. Inputs were reported per kilogram of Hot Standard Carcase Weight (HSCW) which is a standard descriptor in Australia. However, the inputs are inclusive of all stages of meat processing, which included chilling, boning and rendering of co-products. Importantly it should be noted that the definition of HSCW may vary between species and countries.

2.3. Product Yields

Meat processing yields different products depending on the species. Yields may be determined from primary output data, but in the absence of these data the mass of products may be determined from a series of factors. Here we report the yield of products from processing based on primary data and industry averages applicable in Australia. Yield characteristics vary between different breeds of livestock and this can have a large influence on results. As a consequence, important factors such as the dressing percentage and retail yield should be based on data reflective of the supply chain being investigated. In Table 1 we report data relative to the supply of 1000 kg of retail portions. We consider offal sold for human consumption as part of the functional unit, because this product is functionally equivalent to meat from the animal carcase with respect to nutritional characteristics. The mass of unprocessed rendering material and products from rendering are included.

Table 2 provides critical yield factors to convert live weight of animals to the amount of edible meat after meat processing for each species, including the edible fraction for the retail portions, which differs depending on the amount of bone included in the product sold at retail. This will vary depending on the degree of processing (boning), the degree of trimming for excess fat, and the inclusion or exclusion of skin in the saleable portion for chicken and pork. We specify here the edible portion, meaning the mass of product exclusive of bone and cartilage mass which are not easily digested. Importantly, the mass of meat actually consumed may be lower depending on consumer preferences and this would need to be accounted for during the consumption phase. We include the edible fraction here because it differs considerably between the species and therefore is necessary for correctly interpreting the results.

Table 1. Product mass for four meat species relative to 1000 kilograms of retail product

Product description	Product	Chicken meat	Pork	Sheep meat	Beef
Farm-gate product	Live Weight	1712	1618	2255	2375
Intermediate product	Hot Standard Carcase Weight	1216	1229	1060	1307
Wholesale product	Cold Carcase Weight	1179	1177	1017	1267
Wholesale/retail product	Retail cuts	967	906	895	887
	Edible offal	33	94	105	113
Retail portions (retail cuts + edible offal)		***1000***	***1000***	***1000***	***1000***
	Hides	0	0	169	214
	Pet food	17	84	34	30
Rendering material	Unprocessed meat, bone, offal	593	372	800	989
Rendering Products	Protein meal products	154	53	152	257
	Tallow	73	109	156	154
	Blood meal	5	5	14	14

Table 2. Meat processing yield factors for chicken meat, pork, sheep meat and beef

	Chicken meat	Pork	Sheep meat	Beef
Dressing percent	0.71	0.76	0.47	0.55
Chilling loss	0.03	0.04	0.04	0.03
Retail yield	0.82	0.77	0.88	0.70
Edible offal yield (% of LW)	0.02	0.06	0.05	0.05
Edible fraction of retail portions	0.85	0.85	0.76	0.95

2.4. Co-production

The options for handling co-production according to ISO 14044 (ISO 2006) can be divided into two broad approaches, in order of preference.

Methods to avoid allocation:

- Clear subdivision of the system, or
- System expansion (expanding the product system to include the additional functions related to the co-products to avoid allocation).

Allocation:

- Allocation on the basis of physical or biological relationship.
- Allocation on some other basis; most commonly economic (market) value.

In meat processing, sub-division has been proposed as a method of dividing impacts from meat products and rendering material (Ramirez et al. 2011) though this implies that rendering material is valueless and carries no burden from the production system. This is not valid for important co-products such as hides. Where the system can't be divided, system expansion and allocation methods are recommended. In practice, a number of studies that have included allocation in meat processing have applied economic methods (i.e. Ledgard et al. 2010; Milà i Canals et al. 2002; Opio et al. 2013).

System expansion is performed by allocating all burdens to the meat product then deducting impacts associated with the production of substitute products. This method relies on the selection of appropriate substitution products, which are typically the marginal primary products in the relevant commodity market. The ease with which substitution products can be identified varies. Animal protein meals are used in most parts of the world (with the exception of Europe) as an important feed ingredient for poultry and pigs. Animal protein meals provide a high value protein feed with a superior amino acid profile and higher levels of phosphorus than vegetable protein sources. None the less, they can and are substituted with vegetable protein sources, synthetic amino acids and alternative phosphorus sources. In the present study, we simplified the substitution process by using soybean meal and cereal grain (in this case sorghum) to create an equivalent level of energy and protein as provided by the animal protein meal. We recognize this is a simplification and could be improved by including additional synthetic amino acids to address specific deficiencies. Soymeal from Brazil and Argentina were considered the appropriate marginal protein source and the impacts from this soymeal were determined following Opio et al. (2013). We acknowledge that in some regions, animal protein meals may be used for alternative purposes such as fertilizer but animal feeding is a more important activity globally. Tallow is used for many purposes; human food, animal feed, cosmetics and biofuel to name a few. Here we substituted tallow for palm oil, which is considered the marginal oil source globally. Pet food was substituted for soymeal on a protein equivalent basis as many manufactured pet foods are heavily reliant on vegetable protein sources. This substitution overlooks one important element; animal meals provide both protein and an important meat flavor for pet food which improves the palatability of these products. Hence, the substitution is incomplete. The last major substitution product was the raw hide, which was only relevant for sheep and beef in the Australian examples. Raw hides are processed into leather and other co-products; typically at a separate facility. Because raw hides are an intermediate product, no direct equivalent exists. To add to the complexity, the final leather product operates as a primary product in premium upholstery, footwear and apparel markets. While substitutions can be found, they may be inferior in terms of market acceptance, value and in some cases durability. We applied a simplified substitution approach by determining the mass of hides that contribute to the final leather product (20%) and substituting vinyl at a ratio of 4:1 by mass (European Commission 2004) to account for the superior wear characteristics of leather (assumed to be 80 years). The greater mass of vinyl was to account for the higher durability of leather for use as upholstery or apparel. The mass of avoided product associated with meat production is provided in Table 1 while the avoided impacts per kilogram of product are provided in Table 3.

Table 3. Avoided products used in System Expansion (EcoInvent)

	GHG	**Energy**
	kg CO_2-e	MJ / kg
Leather substitute	9.7	215.9
Pet food (avoided soy) [a]	1.6	0.6
Animal protein meal (avoided soy) [a]	5.8	2.1
Tallow (avoided palm oil) [a]	1.5	4.3
Blood meal (avoided soy) [a]	9.7	2.0

[a] dLUC has been included for palm oil and soy production.

A biophysical allocation approach was applied by using the mass of utilizable protein and energy in each product (Table 4) based on Kanagaraj et al. (2006) and Meeker (2006). Meat products were assumed to have a standard proportion of protein (0.2) and fat (0.15) and were adjusted for the edible yield of product (Table 1). Hides were adjusted for the proportion of protein utilized in the leather product.

Table 4. Utilisable protein and energy in primary and co-products from meat processing

	Chicken meat	**Pork**	**Sheep meat**	**Beef**
Retail portions	0.30	0.30	0.27	0.33
Hides			0.24	0.24
Pet food	0.09	0.09	0.09	0.09
Secondary Rendering Products				
Animal protein meals	0.60	0.50	0.50	0.50
Tallow	1.00	1.00	1.00	1.00
Blood meal	0.85	0.85	0.85	0.85

The very high levels of utilizable protein and energy in the rendering products was because the rendering process removes excess moisture in comparison to meat, hides and pet food which are all high moisture products. This led to high allocation fractions to these products (Table 5) which were not meaningful considering the minor importance of these products.

Table 5. Allocation fractions based on utilisable protein and energy in primary and co-products from meat processing

	Chicken meat	Pork	Sheep meat	Beef
Retail portions	0.63	0.67	0.48	0.49
Hides	0.00	0.00	0.07	0.07
Pet food	0.003	0.017	0.006	0.004
Secondary Rendering Products				
Animal protein meal	0.20	0.06	0.14	0.19
Tallow	0.16	0.25	0.28	0.23
Blood meal	0.01	0.01	0.02	0.02

Alternatively, we applied a hybridized method where we divided the rendering process from the system and used the utilizable protein and energy to perform a biophysical allocation on the retail portions, hides and pet food (Table 6), combined with a system expansion approach to handle the products from rendering.

Table 6. Allocation fractions based on utilizable protein and energy for retail portions, hides and pet food

	Chicken meat	Pork	Sheep meat	Beef
Retail portions	0.99	0.98	0.86	0.86
Hides	0.00	0.00	0.13	0.13
Pet food	0.01	0.02	0.01	0.01

Economic allocation was based on the value of products leaving the processing or rendering plant. While rendering is a separate process, many plants surveyed could not provide a separate break-down of energy inputs to rendering as distinct from meat processing. Hence we considered the output of meat processing to be inclusive of rendering. The value of hides was for the unprocessed raw product. While this was in line with the system boundary, it resulted in an inconsistency in the economic allocation method because the products were at a different stage of processing. In agricultural systems, product value is often disproportionately distributed to the wholesale and retail end of the supply chain. Hence, bias can be introduced if economic allocation processes are not performed at the same point in the production system for each product. This will result in lower impacts for less processed products (i.e. leather) and higher impacts for more processed products such as meat and animal meals. An interesting alternative not applied here would be to expand the primary processing system to include tanning of leather products to align the production stages for all products. Economic allocation proportions are provided in Table 7. The value of retail portions was based on the wholesale value of all meat products as reported by the meat processing plants and verified using Australian market data where available (APL 2009; MLA 2013a; MLA 2013b).

Table 7. Economic allocation factors for meat products and co-products from four species

Slaughter Products	Chicken meat	Pork	Sheep meat	Beef
Retail portions	97.2%	97.5%	88.3%	89.1%
Hides	0.0%	0.0%	8.0%	5.7%
Pet food	0.2%	0.4%	0.4%	0.5%
Secondary Rendering Products				
Animal protein meal	1.3%	0.5%	1.3%	2.5%
Tallow	1.2%	1.5%	1.7%	2.0%
Blood meal	0.1%	0.1%	0.2%	0.2%
Total	100.0%	100.0%	100.0%	100.0%

3. Results

3.1. Impacts from meat processing

Meat processing varied substantially between species for all impact categories of interest. Energy and GHG emissions were closely associated, though additional emissions also arose from anaerobic waste treatment systems were utilized. Sheep processing was the most energy and GHG intensive, followed by beef processing. These also used the highest amount of water. In the context of the whole supply chain, meat processing was a significant contributor to energy demand for all species.

Table 8. Meat processing impacts associated with processing four different species, expressed as per kilogram of Hot Stand Carcase Weight (HSCW)

	Units	Chicken meat	Pork	Sheep meat	Beef
Global Warming	kg CO_2-e / kg HSCW	0.42	0.40	0.91	0.80
Fossil energy (MJ)	MJ / kg HSCW	4.34	3.00	8.33	5.81
Consumptive Fresh Water Use	L / kg HSCW	2.43	6.57	7.53	8.75

3.2. Handling co-production

Results without allocation are provided in Table 9 using four different reporting units for comparison. The results for HSCW without processing impacts represent a common approach, where impacts are calculated to the farm gate then divided by an assumed dressing percentage without allocation or accounting for processing impacts. Including processing impacts contributed an additional 3-11% with the greatest proportional increase being for chicken meat and the least being for beef. Results are also presented per kilogram of retail meat without allocation.

Table 9. Greenhouse gas emissions (kg CO_2-e / kg product) from four species reported on an unallocated basis per kilogram of live weight, HSCW and carcase retail yield

	Chicken meat	Pork	Sheep meat	Beef
Live weight	2.5	4.4	6.4	11.7
HSCW (no allocation, no processing impacts)	3.5	5.7	13.6	21.3
HSCW (no allocation to co-products, processing impacts included)	3.9	6.1	14.5	22.1
Carcase retail yield (no allocation to co-products, processing impacts included)	5.8	9.8	22.7	34.3

GHG emissions for four species and four methods for handling co-production are shown in Figure 1. Impacts were lowest for the utilized protein and energy method and highest for the economic allocation method, with the system expansion and the hybrid method being intermediate. The utilized protein-energy method was not considered suitable because the allocation to minor rendering products was very high, resulting in higher impacts per kilogram of co-products than the primary product. The difference was primarily because of the difference in moisture content; the retail products have a high proportion of moisture which increases product mass, while the rendering products are dry (90% dry matter). Economic allocation tended to attribute slightly lower impacts to co-products than the comparative system expansion approach, though these were fairly similar across the species. The hybrid method was effective in allocating impacts between the utilized portion of the meat, hides and pet food while avoiding the errors created by including the minor co-products into this method.

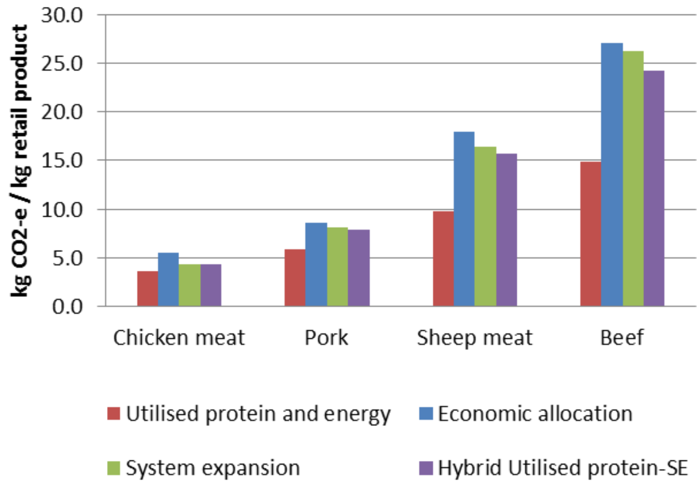

Figure 1. Greenhouse gas emissions for meat products with four alternative methods for handling co-production across four species

The assessment of energy demand (Table 10) followed the same trend as GHG for the allocation method because of the fixed ratios applied to both indicators, but showed strongly divergent results for the SE method for sheep and beef. This was largely influenced by the substitution process for hides, which were replaced by vinyl, which is an energy intensive product.

Table 10. Energy demand from four species reported using four different approaches for handling co-production

	Chicken meat	Pork	Sheep meat	Beef
Economic allocation	21.8	25.0	18.1	20.7
Utilized protein allocation	14.2	17.2	9.9	11.4
System expansion	21.7	24.8	-18.0	-25.5
Hybrid Utilized protein-SE	21.6	24.1	15.7	17.4

4. Discussion

A few Australian studies have found that meat processing contributes a relatively small amount of GHG to the total impacts from meat production, but higher amounts of energy (Peters et al. 2010; Wiedemann et al. 2010; Wiedemann et al. 2012). The contribution is greatest for ‘low impact’ meat species such as poultry where the contribution to GHG was 11% and to energy was 31%. Meat processing is a very important stage in the life cycle of meat products from the perspective of handling co-products. This analysis showed significant differences in the impacts attributed to the meat product depending on whether co-products were taken into account, and depending on which method was used. While it is common in the literature for results to be presented without allocating impacts to co-products, we found this induced a substantial error, over predicting the impacts associated with meat by between 6% and 42% depending on the species and method compared. The use of intermediate reporting units such as carcase weight introduced variation between the species, because of the different final yield relative to carcase weight and because dressing percentages do not take into account the yield of edible offal. For this reason, we preferred a functional unit that represents the retail yield from the animal when the focus is food production. Retail yield is still not uniform, because different species include or exclude different amounts of bone in the retail product and therefore provide a different amount of digestible protein and energy. While accounting for the edible portion of meat is part of the consumption phase of the life cycle, it is useful to report this and take it into account in the interpretation phase, particularly if considering different species. The error induced by reporting results on an unallocated basis related to a number of factors. Firstly, the practice of allocating to carcase weight and retail meat from the carcase, while excluding edible offal products, reduces the apparent product yield. We found no reason to exclude this from the functional unit as it contributes equally to human food supply. Including edible offal in the functional unit reduced the total live weight required for each kilogram of retail product by between 3% (poultry) and 12-13% (sheep and beef).

Of the allocation processes, economic allocation has been most commonly applied to date. Economic allocation resulted in a 6 to 27% reduction in impacts allocated to the meat product depending on species compared to the unallocated process, with the greatest differences coming from ruminants. Across the allocation and system expansion methods, the differences in comparison to the unallocated methods were greatest for ruminant species. This was partly because of the role of edible offal discussed above, and partly because of the handling of the animal hide. In poultry and pigs, animal skin is commonly sold with the retail product, where it becomes part of the consumed product or is thrown away by the consumer. In ruminant species, hides are commonly sent to an additional tanning process to produce leather. Leather is a global benchmark product that functions in the market in a fashion more similar to a primary product than a co-product. It is also a high value product at the wholesale and retail level, though this is not reflected in the raw hide value. Because of the point in the processing supply chain where hides are removed, hides receive a relatively small proportion of the environmental burden compared to the biophysical ‘cost’ of producing hides over the life of the animal. This was evidenced by the higher impacts attributable to hides when applying a biophysical allocation process based on utilized energy and protein in the product. However, we found it difficult to establish a meaningful biophysical allocation process that could cover both the major products (i.e. retail meat and hides) and the minor products from rendering. Similarly, we found it difficult to determine an appropriate system expansion process to handle hides, because they are a raw product rather than a final product. A more thorough analysis is required to assess this by expanding the system to take into account the tanning process also.

We presented here a hybrid method that used the favored biophysical approach to divide impacts between the major products while using system expansion to account for the minor rendering products. While it has been accepted for some time in LCA practice that multiple methods of allocation may be used at different points in the same study, we apply this here for a closely related co-product system coming from the same process. We consider this to be an ideal method for accurately attributing impacts to meat and hides, which are important primary products, while accounting for the lesser volumes of rendering co-products with a more appropriate system expansion method.

5. Conclusion

This paper presents new inventory data, product yields and an exploration of novel methods for handling co-products at the point of meat processing. We suggest here a re-definition of the functional unit for meat products to include the yield of edible co-products that are an important contribution to human food supply. With the re-definition of the functional unit and application of allocation or system expansion methods, impacts attributable to meat products declined by 6-42%. The difference in impacts attributed to the meat product was particularly important for ruminant species, which produced the largest mass of important co-products of all the species. The data and methods presented here provide a starting point for researchers to include this approach for future livestock and meat studies.

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Impacts of land use change on the assessment of water use in grazing systems and interactions with carbon sequestration

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ABSTRACT

The impact of land use change (LUC) and water use was investigated using a a case study of beef production from grazing land. LUC from grassland to forest (afforestation) reduced greenhouse gas (GHG) emissions as a result of carbon sequestration, but also decreased runoff and consumptive fresh water availability compared to pre-LUC conditions. During the transition (LUC period), annualized carbon sequestration (reported as negative emissions) were an estimated –4.9 t CO_2-e / ha.yr. By the end of the transition period (post LUC) run-off declined by 30% (0.1 ML / ha.yr) as tree growth modified soil moisture conditions. Per kilogram of beef, net emissions changed from 10.9 kg CO_2-e / kg live weight (LW) (pre-LUC) to -3.0 kg CO_2-e / kg LW (LUC period). Consumptive water use increased from 305 L / kg LW (pre-LUC) to 632.7 L / kg LW during the LUC period. In the post LUC period, net sequestration was minimal as trees reached maturity, but reduced runoff and higher consumptive water use remained as a permanent change.

Keywords: beef, grazing, LUC, water, carbon, GHG

1. Introduction

It is now widely accepted that land use change (LUC) can be an important contributor to greenhouse gas (GHG) emissions in food production supply chains and is a necessary inclusion for agricultural and food product life cycle assessment (LCA) and carbon footprinting (CF). Land use change can also lead to carbon sequestration (GHG removals) if afforestation occurs. In Australia, efforts have been made to promote tree planting in grazing areas to provide improved production and environmental outcomes (Youl et al. 2006). Afforestation has also been promoted as a means to sequester carbon under a government scheme known as the Carbon Farming Initiative or CFI (Department of the Environment 2014). Accounting for LUC GHG emissions has shown large impacts may be attributed to agricultural systems where LUC occurs, i.e. Cederberg et al. (2011). Other authors such as Schmidinger and Stehfest (2012) have extended the discussion to consider not only the emissions that may have occurred from deforestation, but also the 'missed potential carbon sink' associated with land occupation. This approach treats all land as having the potential to sequester carbon if returned to forest.

Land use change is known to have an effect on water availability (Mila i Canals et al. 2009) though few case studies have considered this. Research in the field of hydrology is unequivocal; increasing vegetation cover will reduce runoff (water yield) while decreasing vegetation cover will increase water yield (Bosch & Hewlett 1982, Brown et al. 2005). Thus, a trade-off exists between net LUC impacts on GHG (emissions or removals) and water yield, and methods proposed to account for the impact of land management on GHG flux (i.e. Schmidinger & Stehfest 2012) should consider broader impacts such as water availability to provide a holistic analysis. In the context of greater global demand for food (FAO 2009), the complex interactions between GHG flux, livestock (food) production and water resource availability require careful investigation to avoid unintended outcomes. Australian grazing land is predominantly classified as grazing of native vegetation (Lesslie & Mewett 2013) and in these systems, livestock graze partly forested pasture lands. In such systems, there is some flexibility to alter the proportion of trees and pasture by promoting seedling establishment from established trees or alternatively, by controlling seedling recruitment. This paper investigates the interaction between these three factors at the local scale for an Australian beef grazing farm where afforestation had taken place as a result of changed farm management, resulting in changed tree density.

2. Methods

2.1. Goal and scope

This study used actual production data from a cattle grazing farm in Queensland, Australia to investigate the impacts of partial afforestation of pasture land. The case study was located in the headwaters of the Murray Darling river catchment of Australia, which has a high degree of competition for water resources. This study examined the impacts of LUC on global warming using IPCC AR4 global warming potentials (GWP) – (IPCC 2007). Consumptive water use was assessed using water balance methods (Bayart et al. 2010). The system boundary included all processes associated with the production of beef up to the farm-gate. The functional unit (FU) was chosen as 'one kilogram of live weight at the farm gate' and results were also reported as totals for the whole farm.

2.2. Inventory data

Livestock production data were collected from farm records over a two year period and were confirmed via discussions with the farmer and site visits in the year 2010 (Table 1). Land use change was initiated on the farm in 2008 as a result of a change in grazing species (sheep to cattle) when the farm was purchased. This resulted in natural regrowth of native woody perennial trees (afforestation) to take place on a 235 ha section of the property. On other parts of the property, regrowth was controlled as part of routine management. The association between cattle grazing and woody thickening has been reported large parts of Australia (Burrows et al. 1998, Burrows et al. 2002). On the case study farm, woody thickening was the result of a change in grazing species from sheep, which graze young trees and limit growth, to cattle which don't graze young trees to the same extent. Grazing livestock were changed from sheep to cattle eight years before the assessment period (in the year 2000), resulting in wide spread establishment of native trees that were suppressed by sheep grazing but not by cattle. While attribution of the impacts of LUC to livestock is generally accepted where livestock are grazed following conversion of forest to pasture, attribution is less clear when changed grazing management results in increased tree growth and GHG removals. On the case study farm, the management change from sheep to cattle was deemed the causal factor enabling establishment of trees, providing a causal association between cattle grazing and changed GHG flux.

The system was assessed over three time periods, before (pre-LUC, prior to year 2008), the LUC period (2008-2037) and the post-LUC period (2037+). The impact of afforestation was modelled over a forward estimate period of 30 years to approximate the time taken for trees to reach maturity. The modelling periods did not account for other possible changes in climate or management over the forward estimate period. Over the reforested land area, livestock grazing was predicted to decline by 10%, corresponding to the decline in grass growth as trees approached maturity. This was predicted to result in an equivalent 10% decrease in livestock numbers and beef production for the farm in the absence of other management changes.

Table 1. Case study farm characteristics

Parameter	Units	Value
Average annual rainfall	mm	661
Land and grazing		
Farm size	ha	2 017
Total area of pasture	ha	1 967
Afforestation area	ha	235
Sheep	%	0
Cattle	%	100
Cropping	%	0
Herd size (cow no.)	No.	212
Water supply		
Farm dam	%	83
Bore	%	0
Creek	%	17
Total livestock drinking water	ML	5.27

Consumptive fresh water use

The water use inventory was developed in accordance with Bayart et al. (2010), covering all sources and losses associated with beef production both in foreground and background systems. Primary sources of consumptive fresh water use for beef cattle production were associated with livestock drinking water, drinking water supply losses, and irrigation. Drinking water for grazing cattle was predicted from feed intake, climate and feed characteristics for the farm using equations from Ridoutt et al. (2012). Losses from farm dams occurred from evaporation, and to a lesser extent seepage (Nathan & Lowe 2012). Losses associated with water supply from farm dams were modelled using farm dam water balances constructed from long term climate data for the farm. Dams and catchment areas were assessed during site visits and were later mapped using aerial imagery. The dam water balances were modelled using a daily time-step water balance using long term rainfall and evaporation data obtained for each region as Patched Point Datasets (DSITIA 2013, Jeffrey et al. 2001). The balance accounted for extractions, seepage and evaporation losses. The dam water balances were calibrated using records of filling and emptying events, determined through discussion with the farm owner. Catchment runoff and dam inflow was modelled using USDA-SCS KII curve numbers (USDA NRCS 2007), with appropriate values determined from site observations of soil type and farming practices. Water use was determined from the difference between water flows leaving the farm in the presence of absence of the farming system. Water modelling took into account the degree of forest cover in prior to LUC (pre-LUC), during afforestation (LUC period) and post-LUC. Runoff predictions were calibrated at the local scale using farmer knowledge of the frequency of runoff events, and against catchment yields for similar catchments. Catchment yields (runoff as a percentage of rainfall) were 6% in the pasture system, declining to 4% in the reforested system after 30 years which was a conservative estimate (Brown et al. 2005), and similar to the estimated whole catchment water yield of 5% (CSIRO 2007). This corresponded to an annualised decline in runoff across the whole farm of 25.4 ML per year over the LUC period and an absolute decline of 31 ML between the pre and post-LUC periods.

Greenhouse Gas Emissions

Greenhouse gas emissions from livestock were determined using methods based on the Australian National Greenhouse Gas Inventory (DCCEE 2012) with the exception of the enteric methane prediction equation, which was based on a regionally representative enteric methane prediction model (Kennedy & Charmley 2012). All impacts associated with energy demand were included, based in the inventory of farm purchases (data not shown). No additional energy use associated with afforestation because this occurred naturally. Hence, emissions intensity associated with livestock emissions and energy use remained the same. Net LUC emissions were determined from Fensham and Guymer (2009) for sub-humid, Eucalypt woodland, which was assumed to have a net sequestration 40 t C ha and a total sequestration of 34,466 t CO_2-e over the 30 year period. As an annualized rate, this resulted in sequestration of 1149 t CO_2-e per year for the farm.

3. Results

Total beef production and GHG emissions are presented for the three time periods in Table 2. Land use change resulted in net removals from sequestration of -14.4 kg CO_2-e / kg LW, which resulted in a net negative emission of -3.5 kg CO_2-e / kg LW for all beef sold from the farm over a thirty year period with all emission sources taken into account. However, after the forest reached maturity, estimated total beef production declined by 10% and estimated net emissions returned to levels similar to the pre LUC emissions.

Table 2. Greenhouse gas emissions and sequestration associated with land use change reported over three time periods

Time period	Beef production	GHG	LUC GHG	total GHG	Emissions intensity
	total kg LW	t CO2-e	t CO2-e	t CO2-e	kg CO2-e / kg LW
2008 (pre LUC)	88,649	966	-	966	10.9
2008-2037 (LUC period)	82,443	899	- 1,149	- 250	-3.0
2037+ (post LUC)	79,784	899	-	899	11.3

Consumptive water use over the same time period changed in response to changing livestock numbers and changed vegetation cover. This resulted in a net decline in direct consumptive water use of 2% during the LUC period and 3% after the LUC period ended. This change in direct consumptive water use was in response to lower drinking water requirements for the smaller herd. Importantly, the decline was not a linear response to reductions in livestock numbers and production. This non-linear response was because the reduction in drinking water requirements had a negligible impact on water supply system losses from farm dam evaporation. In the present case study, the farm manager did not respond to reduced livestock numbers by changing either the number or size of dams. This is to be expected, because changing the number or size of dams would require considerable capital expenditure without providing any benefit to the farm. Hence, the supply losses associated with farm dams occur despite changed livestock numbers.

The large changes in water were the result of changed runoff conditions, which may be termed an indirect water use for the system. When attributed to the livestock system, this resulted in an additional 308 L / kg LW. This additional indirect water use continued as a permanent change to local hydrology after the LUC period finished, resulting in a large, long-term increase in consumptive water use that would be reduced only after another LUC event to reduce vegetation.

Table 3. Change in consumptive water use in response to land use change reported over three time periods

Time period	Beef production	Consumptive water use	Change in runoff (LUC)	Total water	Water use / kg beef
	total kg LW	ML	ML	ML	L / kg LW
2008 (pre LUC)	88,649	27.1	-	27.1	305.2
2008-2037 (LUC period)	82,443	26.8	25.4	52.2	632.7
2037+ (post LUC)	79,784	26.2	31.1	57.3	718.4

4. Discussion

There is a global imperative to reduce greenhouse gas emissions from livestock production, and emissions from land use change are understood to be a substantial contributor to total emissions (Opio et al. 2013). One possible option is to promote afforestation of grazing areas. The environmental impacts from this type of land use change are likely to be improved in some instances (i.e. improved biodiversity outcomes) but there are other impacts that must be addressed, such as the decline in runoff. In water footprint terms, this amounts to a change in the proportion of green and blue water use within a system, because runoff (blue water) declines in response to increased vegetative cover and subsequent evapo-transpiration (green water). This relationship between vegetation and runoff is well established at the global level (Bosch & Hewlett 1982, Brown et al. 2005). Mila i Canals et al. (2009) identified the need to address changes in runoff caused by alterations in the balance of green and blue water within a system. However, the choice of reference system is debatable. In the present example, we chose the land cover in 2008 as the reference. At this time, the land was open pasture used for grazing. However, if the reference system was considered the natural vegetation at the site prior to land clearing (which occurred around the 1900's for this region) then afforestation has a very different effect on sequestration and net water use. Such matters are not semantic, they demonstrate the importance of such methodological choices on the interpretation of results. In Australia, the relationship between vegetation cover, runoff and subsequent water available for competitive users is carefully monitored, and programs promoting afforestation must account for the impact on reduced runoff (NWC 2011). It would be clearly beneficial for LCA research to address this rather than considering LUC in the context of GHG emissions only. The implications of this research on deforestation are also apparent; local scale deforestation will result in increased runoff and greater water availability (Bosch & Hewlett 1982, Brown et al. 2005). This relationship has been assessed as a means to increase runoff and alleviate localized water stress in Australia (Li et al. 2011). In the present case study, increasing carbon storage in vegetation had a relatively short term effect (30 years) but resulted in a permanent decrease in water availability and beef production. The negative impact of lower water availability in river systems impacts both the environment and competitive water users such as irrigators elsewhere in the river system. It is a legislated requirement that afforestation projects funded under the Australian government CFI program account for reduced runoff, possibly via acquisition of irrigation water licenses to account for water consumption (NWC 2011). This aligns with an approach that considers this water an additional consumptive use in the LCA context. The focus on this paper has been to consider the localized impacts of this change only, rather than a full scale consequential analysis. The study only took into account one additional impact, consumptive water use, and further study should incorporate the impact on a broader range of potential impacts, including soil degradation and water quality. A full system consequential analysis is also required to understand the impacts of changes in water availability and beef supply. For example, reduced beef production in Australia (one of the world's two largest beef exporters –FAO (2011) may induce beef production in other regions where deforestation is a risk. Secondly, reduced water yield will constrain water supply to local river systems, causing stress to aquatic ecosystems and reducing irrigation water supply, with the latter having ongoing impacts on food and fibre production.

5. Conclusion

Afforestation is a plausible LUC option to remove atmospheric carbon dioxide via carbon sequestration in woody perennials. In a mixed grazing system, GHG removals from afforestation may be triggered by beef grazing, and as a result could provide reduce net emissions from beef production. However, such improvements can only be sustained for a limited period of time until trees reach maturity. After this time, the production system was estimated to return to a net GHG source, with lower overall productive capacity. Importantly, LUC resulted in reduced water yield which may place additional stress on water catchments, compromising aquatic ecosystem function and reducing water supply for irrigation and food production. This highlights the important trade-offs that exist when assessing LUC, particularly with respect to GHG emissions and water yield, and consideration of a broader suite of impacts such as impacts on soil condition would be beneficial. Detailed consequential modelling is required to explore the broader implications of these findings, accounting for environmental impact and the effect on global food supply. In the context of greater global demand for food and the environmental challenges such as climate change, this research should be seen as a priority.

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A simple approach to land use change emissions for global crop commodities reflecting demand

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ABSTRACT

A development of the top down method for accounting for all direct and indirect land use change emissions (LUCE) is presented, which reflects the relative rates of global crop expansion. It uses crop production and area data from FAOSTAT to derive expansion rates, reflecting global demand. Crops that drive land use change more, e.g. soy thus receive a heavier burden than other. It is thus more equitable than the original top down method. Alternative sources of values for LUCE and connected agricultural area are addressed. It still represents a method with relatively low computational and data demands.

Keywords: greenhouse gas emissions, land use change, crops, top down, soy

1. Introduction

Greenhouse gas (GHG) emissions (GHGE) from land use change (LUC) have a large effect on life cycle assessments (LCA) of food production, especially from livestock products (e.g. Audsley et al., 2009, Leinonen et al., 2013, Leinonen et al., 2014, van Middelaar et al. 2013, Meul et al., 2012, Cederberg et al., 2011). The relative importance is even greater in life cycle based studies such as determining product carbon footprints or if calculating GHGE from national or regional food consumption.

Determining the GHGE from land use change (LUCE) is relatively straightforward if the history of the land parcels is well known and if the initial conditions of soil and biomass carbon densities are known. This is not, however, always the case, neither does this address the consequences of indirect LUCE.

Current approaches to determine direct LUC GHG emissions (direct LUCE) include the UK publically-available specification PAS 2050, which started with one method as (BSI, 2008), and was later revised in (BSI, 2011). In BSI (2008), the default position if land history was not known was to assume the worst case scenario of LUCE: deforestation. This was revised in BSI (2011) to apply the weighted average of direct LUCE from the originating country, although it did not specify exactly how this should be applied. If dealing with a major commodity, such as soy, the data requirements are potentially very high, given the large areas of land and multiple countries. The method is based on the premise that one land use is succeeded by another, e.g. forest to arable. It does not address land that may have been deforested, then abandoned and regenerated towards forest. Fearnside (1997) addressed this with the concept of “net committed emissions”, which also requires much detailed data to be effectively applied. This approach was also used by Cederberg et al. (2011). These methods do not really address indirect LUC fully.

A radical alternative, the “top down” approach was developed by Audsley et al. (2009), revised by Vellinga et al. (2013) for the Dutch feed industry and compared with other approaches by van Middelaar et al. (2013). In the top down” approach, all global direct and indirect LUCE are applied uniformly to all economically connected agricultural land. The underlying principle is that commodity demand will be met by the world market, so that land expansion operates collectively in response to demand. LUC in one area may result from changed demand in any other, whether direct or indirect.

Data needs are relatively modest and can be derived mainly from FAOSTAT land use and production statistics. This is coupled with an estimate of global LUCE, of which several possible values exist. These differ in response to inclusion or not of emissions from source such as nitrogen transformations when soil C stocks are depleted. Values for total global LUCE range from 3.3 Gt CO_2e/year (DeFries, et al. 2002), through 4.94 Gt CO_2e/year (Audsley, et al., 2009) to 8.49 Gt CO_2e/year (Olivier, et al., 2005). Vellinga & Van Middelaar (2011) used 5.62 Gt CO_2e/year, as a result of averaging 14 different sources.

Another factor that affects the overall result is how much of the global agricultural area should be considered to be connected. Audsley, et al. (2009) used 3.47 Gha, whereas Vellinga & Van Middelaar (2011) estimated it as 4.9 Gha. Differences in the accounting procedure are evident.

A limitation of the original method is that all crops are considered to be “equally guilty” and carry the same burden per ha, whereas soy expansion by area exceeds other crops (e.g. 1.75 more than maize). In an attributional approach such as PAS2050, crops like soy typically have a high LUCE burden, while European wheat would have none. The reality is somewhere in between. This study presents an enhancement of the top down method, which addresses the relative expansion rates of global crop areas. It is thus still (a) relatively simple to apply, (b) responds to changes in commodity demands (hence reflecting economic drivers for LUC) and (c) avoids double counting. As before, indirect LUCE are also included.

2. Methods

Choices in the approach were compared that addressed calculating the rate of change of crop areas, calculating the global agricultural area and the global LUCE. The core feature, however, was including the rates of crop expansion.

Six measurements are considered in this study including the range (4.94, 5.62, 5.8 and 8.1 Gt CO_2e/year).

2.1. General approach

The world’s top 25 crop commodities (by weight produced) were analyzed individually. All other crops were treated as one and grassland as another crop. The top 25 commodities accounted for 85% production on a fresh weight basis or 74% of cropped area. Production and crop area data came from FAOSTAT (2014).

2.2. Calculating land area changes

Annual rates of area expansion (ARE) for the top 25 crops were determined using time spans of 3, 5, 7, 10, 15 and 20 years, with the last year being 2010. 20 years is the period used by the IPPC and in PAS2050 used to capture the bulk of LUCE, although in reality, these may continue beyond 20 years.

Three approaches were tried by Dominguez (2013). The first was simple difference between the final year under scrutiny (A_F) and the initial year (A_i), Equation 1.

$$ARE = \frac{A_f - A_i}{T_f - T_i} \qquad \text{Equation 1}$$

The second used interval proposed in PAS2050-1:2012 interval, in which the average of three years is used in order to reduce fluctuations (Equation 2).

$$ARE = \frac{\left(A_{f-1} + A_f + A_{f+1}\right) - \left(A_{i-1} + A_i + A_{i+1}\right)}{3\left(T_f - T_i\right)} \qquad \text{Equation 2}$$

The third method was simple linear regression. Dominguez (2013) compared the three methods and found similar results, but regression has the advantage of including an estimate of uncertainty. Hence, it is the only method presented here.

The expansion of the other 137 smaller crops was simply derived by the difference between global harvested areas and those of the top 25 crops and treating them as one lumped crop.

The rate of expansion of grassland required screening out the area used by subsistence graziers, e.g. those in sub-Saharan Africa. Audsley et al. (2009) filtered out countries that did not fulfil the following three criteria. (1) Producing less than 0.5% of global production of meat from cattle, sheep and goats. (2) Importing less than 0.5% of globally produced meats. (3) Exporting 0.5% of globally produced meats. Hence, the area expansion of grass only included that from the countries that met all the 0.5% thresholds.

The sum of both crop and grass expansions gave the total annual net rate of expansion (ANRE) for a given period. This is the area that commercial agriculture varied annually, expanding or contracting according to global trends.

The next step is to obtain the proportion of each commodity $Ppt_{(ARE_c)_t}$ in the ANRE.

$$Ppt_{(ARE_c)_t} = \frac{(ARE_c)_t(100)}{ANRE_t} \quad \text{Equation 3}$$

$(ARE_c)_t$ is the annual rate of expansion for a commodity c in a period t, $ANRE_t$ is the annual net rate of expansion in a period t.

2.3. Normalization

The question of dealing with declining crops areas was addressed through normalization. All expansion rates (as proportions of the total) were made positive by adding the smallest integer possible, i.e. 1. Each rate was then divided by the mean expansion rate to give the normalized values for each crop (NV_C), such that the sum of all expansion rates would still sum to the total. These normalized values were then used to scale the total estimate of LUCE for all crops. This was applied because the general trend for the cropped area is expansion and that some "responsibility" should be held by all major crops. The main effect was thus to cause all major crops to incur a portion of global LUCE.

2.4. Baseline emissions from land use change

The baseline emissions from land use change are the LUCE per unit of agricultural land as used by Audsley et al. (2009) and by Vellinga et al. (2013). Two terms are needed for the top down method: the global agricultural area to be considered (Table 1) and the global estimate of LUCE (Table 2). These represent estimates of cumulative LUCE from agriculture.

Table 1. Values used for global agricultural areas harvested

Value, Gha	Source
3.47	Audsley, et al. (2009)
4.90	Vellinga and Van Middelaar (2011)
2.88	Method here studied, includes grassland area from screening

Table 2. Values used for global land use change GHG emissions

Value, Gt CO_2e/year.	Source	Comments
3.30	DeFries, et al. 2002.	
4.94	Audsley, et al., 2009.	58% CO_2e from IPCC 2007 AR4 dedicated to commercial agriculture
5.62	Vellinga and Van Middelaar 2011.	
5.80	Vellinga and Van Middelaar 2011.	Including effects of soil degradation
8.10	Houghton, 2003.	Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850-2000
8.49	Olivier, et al. 2005.	

The value of 1.42 t CO2e/ha derived by Audsley et al. (2009) is simply the result of dividing the global land use change GHG emissions (4.94) by the global agricultural areas harvested (3.47).

3. Results

3.1. Land use change 1990 to 2010

Crop areas expanded from 1990 to 2010 at an average rate of 7.0 Mha/year, with a considerably greater expansion rate from 2002 (Figure 1). The overall increase in cropped area was 12% of the 1990 value. In contrast, grassland expansion increased up to 1996 and declined to 2010, at 1% below the 1990 area. This is a net effect with some grassland being created from deforestation and some being lost to cropping. The overall agricultural expansion is clearly dominated by crops. Linear regression accounted for 86% of the variance in crop area expansion from 1990 to 2010 and 92% from 2001 to 2010.

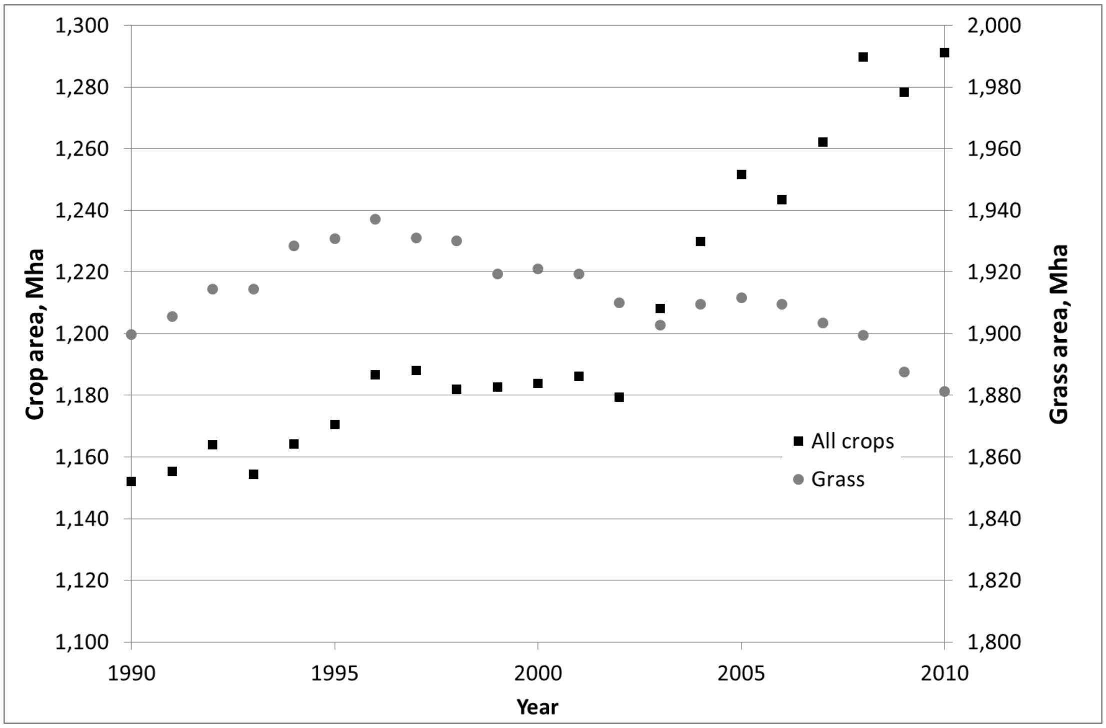

Figure 1. Changes in annually recorded global crop and grassland areas from 1990 to 2010 using data from FAOSTAT and screening out economically isolated grasslands.

3.2. Rescaled land use change emissions 1990 to 2010

When crop area expansion is broken down, it is clear that soy dominates, with maize following at about 55% of the soy rate (Table 3). Rice and rapeseed expanded at about 25% of the rate of soy from 1990 to 2010. In contrast, the area of wheat, the main grain consumed in the UK, has decreased at about 15% of the rate of increase of soy. The effect of normalization is to increase the LUC impact of soy by 36% compare with the single top-down value of Audsley et al, (2009) and to decrease barley by 21% (Table 3). This is a factor of 1.7 between crops in the top 25 of global production and clearly reflects a major difference in demand. Onion is the crop that lies closest to the mean.

These scalars thus increase the LUCE factor per ha of soy from the single value of 1.4 to 1.9 t CO_2e/ha and to decrease the values for wheat and barley to 1.3 and 1.1 t CO_2e/ha respectively. One value is given for all other crops together for convenience, but a separate value should be determined for any other individual crop of interest.

Table 3. Rates of expansion of crops from 1990 to 2010, normalized scalar for each crop and revised LUCE emissions. Crops are ordered by global production in 2010

Commodity	Rate of expansion, kha/year	Normalized scalar of LUC impact	Revised value for LUCE by crop from the single value of Audsley et al, (2009), t CO_2e/ha
Soybeans	2,502	1.36	1.94
Maize	1,431	1.20	1.71
Rice	616	1.07	1.53
Rapeseed	593	1.07	1.53
Oil palm fruit	488	1.06	1.50
Vegetables, fresh	443	1.05	1.50
Sugar, cane	315	1.03	1.47
Cassava	168	1.01	1.43
Onions	110	1.00	1.43
Tomatoes	98	1.00	1.42
Bananas	89	1.00	1.42
Watermelons	79	1.00	1.42
Coconuts	71	1.00	1.42
Cucumbers	47	0.99	1.41
Oranges	39	0.99	1.41
Potatoes	38	0.99	1.41
Cabbages	30	0.99	1.41
Grapes	-26	0.98	1.39
Sorghum	-47	0.98	1.39
Sweet potatoes	-52	0.98	1.39
Apples	-54	0.98	1.39
Cotton	-65	0.97	1.38
Sugar, Beet	-227	0.95	1.35
Wheat	-373	0.93	1.32
Barley	-1,228	0.79	1.13
Other crops	1,887	1.25	1.78
All crops	6,972	1	1.42

3.3. Effect of choice of baseline LUCEs and different estimated agricultural areas dedicated

The choice of what values to use for global land use change emissions and connected agricultural area has major effect on the results (Table 4). The potential baseline values range from 0.67 to 2.95 t CO_2e/ha: a range of 4.4 to 1. However, the more recent review by Vellinga and Van Middelaar (2011) seems likely to give the best estimate for global land use change emissions. The identification of connected agricultural areas depends on a degree of arbitrariness in identifying disconnected grasslands. This study applied the same broad approach as Audsley et al. (2009), but with more recent data and by applying more than one test. It is thus more discriminating.

Table 4. Baseline land use change emissions resulting from combining six sources of land use change emissions with three sources of the areas of connected agricultural activity. Results are in t CO_2e/ha.

		Harvested area data source		
		Audsley el al. (2009)	Vellinga & Van Middelaar (2011)	This study
Global LUC emissions data source	Audsley et al. (2009)	1.42	1.01	1.72
	Vellinga & Van Middelaar (2011)	1.62	1.15	1.95
	DeFries et al. (2002)	0.95	0.67	1.15
	Houghton (2003)	2.33	1.65	2.81
	Olivier et al. (2005)	2.44	1.73	2.95
	Vellinga & Van Middelaar (2011), including soil degradation.	1.67	1.18	2.01

3.4. Effects of time horizon on emissions

The time horizon used for calculating rates of change of crop areas includes market influences coupled with the technical change of generally increasing yields, which increase at a lower rate. There were evident differences over the time scales of 3 to 20 years, e.g. with the factor for soy being 50% larger over 15 than three years (Table 5). Overall, the effects of changing the time scale were relatively small, given the coefficients of variation over the six time periods analyzed. This is particularly the case for onions: the crop that was closest to the mean. The choice of time is arguably arbitrary, but it is rational to use a relatively long period to avoid short term influences.

Table 5. Range of LUCE values for emblematic crops using different time horizons to obtain the rates of change of crop area. Results are in t CO_2e/ha and use the baseline LUCE of Audsley el al. (2009). Onion is the crop closest to the mean.

	Time period of area change analysis up to 2010, years							
Commodities (integers show order in 20 year analysis)	20	15	10	7	5	3	Mean	Coefficient of variation, %
1. Soybeans	2.2	2.5	2.0	2.1	2.3	1.6	2.1	14%
2. Other crops	2.1	2.3	2.1	2.3	2.4	1.6	2.1	15%
3. Maize	2.0	2.2	2.0	2.3	2.6	1.5	2.1	17%
4. Rice	1.8	1.9	1.8	2.0	2.1	1.5	1.9	12%
6. Oil palm fruit	1.7	1.9	1.7	1.8	2.0	1.5	1.8	9%
10. Onion	1.6	1.8	1.6	1.7	1.9	1.5	1.7	7%
25. Wheat	1.5	1.7	1.7	2.0	2.1	1.5	1.7	13%
26. Barley	1.3	1.5	1.5	1.5	1.5	1.4	1.5	6%

4. Discussion

The top down approach offers a method of including both direct and indirect land use change GHGE in analyses, especially for animal diets or addressing national dietary consumption. This study indicates how the approach can be developed to address the relative expansion rates of crops at a global level and hence overcome a perceived limitation of the original method, i.e. all crops are equally “responsible” for land use change. It is evident that the global demand for soy outstrips all others and this approach accounts for that. Crop area expansion is effectively a measure of demand, although tempered by the generally increasing annual crop yields. Thus, where technical effectiveness is greater in increasing yields, the impact in area expansion is reduced. It is still relatively simple method to apply, with much lower disaggregated data needs than if trying to apply the bottom up approach that is implied by adherence to PAS2050 or any similar procedure for a particular crop.

There is opportunity for debate about values of some of the terms used, e.g. time scale, connected area of agricultural land and the global GHGE from land use change. The time scale needs to be sufficiently long to avoid short term fluctuations, but not so long as to miss global market trends and a scale of seven to ten years seems to be appropriate. The connected area of agricultural land is one that could be explored further by scrutinizing the trade in arable commodities to that including any subsistence farming is avoided. One problem that arose in dealing with grassland area changes is that of countries changing borders through major political transformations, e.g. the breakup of the Soviet Union. This presented some obstacles in determining whether grassland should be included or not over the time series.

Vellinga et al. (2013) reviewed the data sources on global land use change emissions and their assessment of the most suitable term seems a reasonable choice. Its continued application also makes analyses compatible with the Dutch *FeedPrint* approach.

5. Conclusion

A development of the top down method for accounting for all direct and indirect land use change emissions is presented, which reflects the relative rates of global crop expansion. This, in turn, reflects global demand and so puts a heavier burden on those commodities that are most dominant in driving land use change, e.g. soy. It is thus more equitable than the original top down method. It represents a method with relatively low computational and data demands.

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Environmental performance of traditional beer production in a micro-brewery

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ABSTRACT

A life cycle assessment of traditional English beer brewed in a microbrewery has been made with blue water, cumulative energy and greenhouse gas emissions as used as measures of environmental performance. The boundary was delivery at the point of consumption. Cumulative energy demand was 4.4 MJ/l, greenhouse gas emissions were 0.38 kg CO_2e/l and blue water volumetric consumption was 3.4 l/l. The main burdens arise from raw material production (especially barley and then malt production), brewing operations, delivery and with waste management relatively small but not negligible. Other studies are not fully comparable, but these results are in the range reported for other beer types, countries and boundaries. Opportunities for environmental performance are limited by the small scale of a microbrewery. More detailed auditing is needed to establish what could be achieved. Existing water consumption appears to be relatively low. Solar generated electricity could be of benefit.

Keywords: Beer, Brewing, Micro-brewery, Water, Carbon footprint

1. Introduction

Beer (or ale) is the traditional alcoholic beverage in Britain. It is brewed in a different way from the globally widespread Pilsner type beer, ("lager" in the UK). Traditional beer (also known as "real ale") is brewed at a higher temperature than lager, is flavored with fertilized hops and is matured and served with live yeast in the casks (barrels), so that secondary fermentation provides the CO_2 when consumed. Lager is usually pasteurized, filtered and gassed with compressed CO_2. The LCA of lager has been studied in Estonia (Talve, 2001), Greece (Koroneos et al., 2003) and Italy (Cordella et al., 2008). The carbon footprint of a bottled ale was determined in the USA by Climate Conservancy (2009). Hotspots from these studies included crop production and processing, brewing and bottling. The UK is second largest brewer and consumer of beer in the EU, after Germany and the UK brews about 4,500 ML beer annually. The UK is the largest producer of cask beer in Europe. Brewing uses much energy and water. Parts of brewing's environmental performance have been studied in isolation in the UK, this is the first LCA and applies to cask rather than bottled beer.

2. Methods

Activity data were obtained from a local microbrewery on brewing operations, resources and fuel for distribution. Crop production was taken or derived from Williams et al. (2006). Energy and water requirements for malting came from the industry. Life cycle inventory (LCI) values for energy carriers and waste management came from the ELCD (2013). LCI values for minor ingredients came from Cranfield-derived data or were estimated by analogy. Co-product (spent brewers' grains) valuation was made by system expansion. The focus was energy use, greenhouse gas (GHG) emissions (GHGE) and water use.

2.1. Goal and Scope definition

The overall goal was to evaluate and determine the energy and water use and greenhouse gas (GHG) emissions (GHGE) as indicators of the environmental impact of the production of beer at a microbrewery and hence to assess its current environmental performance.

2.2. Functional Unit

The functional unit is one liter of "Shefford" beer (3.8% alcohol by volume), the main product, delivered to the point of consumption in pubs and to other distributers, in 2013. It is a draught beer often sold in pubs to the local community and other breweries. It is unpasteurised and unfiltered and so has also a short shelf life. Delivery was in casks that are cleaned and reused for many years, not bottles.

2.3. The system boundary

The product system includes all the life cycle steps from primary production (and associated inputs) through crop processing, brewing, waste management and delivery to outlets. Barley is the main ingredient, which is then malted (steeped, germinated and kilned) to produce malt. Hops are used for flavouring and are grown, harvested, dried and packaged. Brewing uses thermal and electrical energy, water and some minor inputs. Beer is matured in large tanks, casks are filled, the stored for about two weeks for the beer to mature and finally delivered. Wastewater from cleaning etc. is treated in the public sewerage system, spent hops and other wastes go to landfill and most spent brewers grains are mainly used for animal feed (with an estimated 5% sent to landfill).

2.4. Data sources and exclusion criteria

Primary data on the brewery activities, sourcing materials and beer delivery came from the brewers. These were derived from the records of six months brewing. An existing life cycle inventory (LCI) was used for barley (Williams et al., 2006) and one was extensively adapted for hops from oilseed rape (Williams et al., 2010). Energy and water use in malting came from the industry. LCI data for other minor ingredients came from analogues that were developed by Williams et al. (2006) or the literature. Energy use and waste management LCI data were taken from the European Life Cycle Database (ELCD 2012).

All GHGE were expressed on the 100 year basis using based on the IPPC (2007) coefficients. Water use was limited to the volumes of abstracted water used in malting, brewing and cleaning. Blue water used in energy production was not considered, but is thought to be much lower than these quantified sources.

Minor components that were likely to contribute less than 1% to any impact were excluded, e.g. yeast, which may be reused between brews. Fixed assets, like buildings, brewing equipment and casks were excluded.

2.5. Allocation methods

Economic allocation between barley grain and straw was as in Williams et al. (2010). Brewers' grains, when used as animal feed, were assumed to have the same feed value as the milling co-product, wheatfeed, on a dry matter basis, so that the avoided burdens method could be applied. No allocation was required if grains were sent to landfill.

3. Results

3.1. Resource use

The resource use in the production of one batch 1 l of Shefford Bitter beer is given in Table 1 and transport data in Table 2.

Table 1 Resource in the production of one batch (1475 l) of Shefford Bitter beer

Material input	Unit	Quantity
Malt	kg	0.15
Hops	kg	0.0020
Yeast	kg	0.0028
Water	l	3.39
Gypsum($CaSO_4$)	kg	0.00017
Calcium carbonate	kg	0.00017
Lactic acid	l	0.00037
Electricity	kWh	0.11348
Natural gas	kWh	0.12
Sodium hydroxide	l	0.00068

Table 2 Delivery distances and transport types of main raw materials

Product	Transport method	Distance, km
Malt	Bulk lorry	225
Hops	Bulk lorry	572
Hops	Large sea ship	15849
Ancillary material	Medium sized lorry	200
Packaging	Medium sized lorry	200

3.2. Burdens of production and delivery

The cumulative energy demand was 4.4 MJ/l, GHGE were 0.38 kg CO_2e/l and blue water volumetric consumption was 3.4 l/l.

The breakdown of burdens (Figure 2) shows that brewery activities itself dominates energy use (41%) with production of raw materials at 28%, transport at 26% and waste management at 5%. Almost all of the material production was for malt (including growing barley). Thermal energy and energy for temperature control were the most important aspects of the brewing stage. Most of the transport energy was for delivering the beer itself.

The distribution of GHGE was broadly similar with 28% from brewery activities, 30% from raw materials, 20% from transport, but waste management at 20%. Differences are mainly because of emissions of gases like nitrous oxide in agriculture and wastewater treatment.

Blue water use was mostly in the brewery activities (87%) and the rest from malting. The blue water in the brewery is split with 24% going into the beer itself and 63% as process and cleaning water.

In the brewery operations, most energy use is as electricity (73%) and thermal energy (natural gas) the remaining 27%. About 60% of the electricity is used for temperature control of the beer while it matures and awaits distribution.

In waste management, the treatment of brewery effluent in the local public sewerage system is the main process and has a magnitude of about eight times the solid waste management. Net solid waste management has a small negative impact owing to the use of brewers' grains for animal feed. If for practical or economic reasons, brewers' grains are sent to landfill, the cumulative energy use and GHGE increase by 4% and 5% respectively.

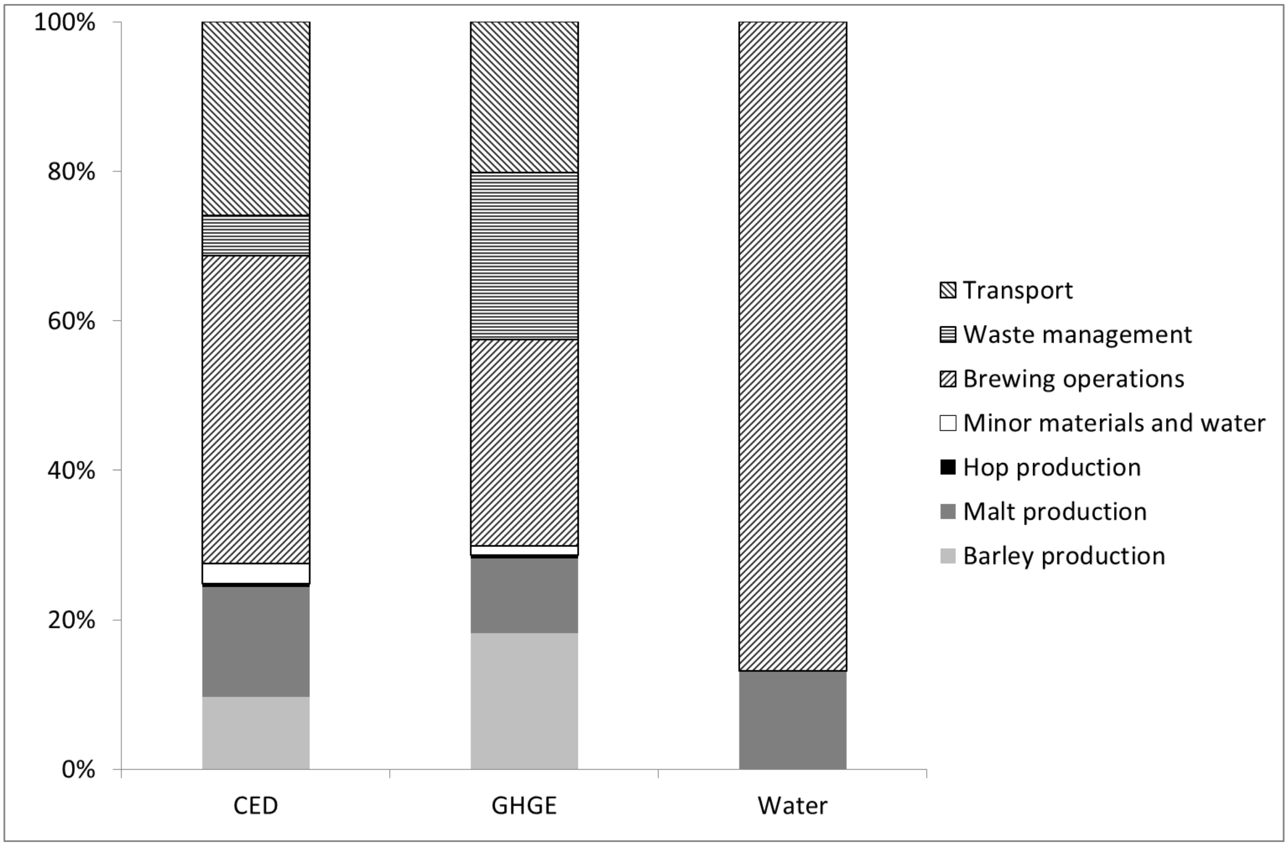

Figure 1 Breakdown of burdens of producing and delivering beer

3.3. Additional transport

Some beer gets sold on to other distributors or breweries to be sold in the free trade or as "guest beers". Extra transport is needed for delivery. The additional burdens caused by this are linearly related to delivery distance. The increase in energy use and GHGE depends also on vehicle type and the effect is estimated in Table 3. Guest beers can readily be found over 500 km from the brewery, which would increase burdens in the range 12% to 40%, depend on vehicle type.

Table 3. Increase in energy use and GHGE at the point of consumption caused by increasing delivery distance by 100 km.

Vehicle type	Cumulative energy demand	GHGE
Bulk lorry transport	2.6%	2.1%
Medium sized lorry	5.8%	4.7%
Small delivery vehicle	8.7%	7.0%

4. Discussion

This is the first LCA of traditional British beer production. It provides an interesting contrast with that of continental styles of beer. It is also applied to a micro-brewery, which will use similar recipes for brewing as larger breweries, but other features will be different owing to economics of scale.

4.1. Opportunities for improvement

The three main stages of raw material production, brewing and delivery dominate most aspects of the burdens, although waste management has a larger relative role in GHGE. A small brewery such as this has little control over raw material production and waste management, especially of effluent. Control over brewing and temperature controlled beer storage in under brewery management control, but these tend to be more constrained in micro-breweries owing to their small scale.

Water use in brewing was relatively low at 3.4 l/l. The brewers of Europe (2002) reported that an efficient brewery uses 4 to 10 l/l and Goldammer (2008) noted that some brewers even uses more water, particularly the small breweries. In this case, there seems to be relatively little scope for improvement, but a more detailed audit could indicate possibilities.

A larger brewery would have its own effluent treatment plant, which if new, could minimize emissions and make positive use any upgrading technologies possible. While this brewery uses heat exchange, there may well be more opportunities for general heat recovery in a larger scale operation. Further, larger scale operations tend to incur lower heat losses. Larger breweries can also capture the CO_2 produced by fermentation for use in its own products, e.g. carbonated beers. This brewery produces no such products.

While the brewers' grains are usually used for animal feed, this does not always occur for practical reasons, such as the farmer having other priorities or not enough animals. An opportunity that is more open to larger brewers is using low grade heat to dry grains so that transport costs are reduced and the market becomes wider.

With a large amount of electricity being consumed, there could be opportunities for using the roof to mount solar panels hence to generate low emissions electricity.

The transport of beer is energetically intensive and this brewery is constrained by having one delivery vehicle for all goods. The delivery routes are not complex and need little optimizing. A larger operation can obtain environmental benefits from a delivery fleet such that loads and vehicles can be better matched. One distinct benefit of the delivery (and subsequent serving) system is that the weight of packaging overheads of cask beer (2.5%) is much smaller than for bottled beers (about 60%).

4.2. Data quality

The activity data was of high quality with good access to data from the brewery and a major maltster in England. The LCI data for processes such as waste management, from the ECLD, are of very high quality, although the effluent treatment data set is generic rather than specific.

A formal uncertainty analysis was not undertaken, but the overall uncertainty of the energy use and GHGE would be in the order of 10%, based on studies undertaken by Wiltshire *et al*., 2009.

4.3 Comparison with other studies

Other studies have inevitably used different boundaries, data sources, data inclusion and beer types, hence all comparisons require great caution.

Cordella et al. (2008) reported a range of energy consumption for Italian lager beer of 3.14 to 5.2 MJ/l, which is similar to the present study at 4.4 MJ/l. Talve (2001), in Latvia, only included electricity in energy use and found much lower value of 0.48 MJ/l. Koroneos et al (2003), in Greece, reported energy use of 0.69 MJ/l, of which beer production was only 6%. It seems to be a considerable underestimate or very efficient processing.

In contrast Koroneos et al (2003) reported GHGE of 754 kg CO_2e/l, compared with 0.38 kg CO_2e/l in the present study. Their value seems to be barely credible and some error seems likely, e.g. unit conversion, such that the real value was likely to be 0.75 kg CO_2e/l. Talve (2001) reported an order of magnitude lower GHGE of 0.065 kg CO_2e/l, but this appeared to omit features like N_2O and CH_4 emissions from agriculture, at least. The Climate Conservancy (2009) calculated carbon footprint for "Fat Tire Amber Ale" in the USA up to and including retail and use as 3.2 kg CO_2e/l. The study is very detailed and covers almost activities including business travel. Making an estimate of equivalent stages to those of the present study suggests GHGE of about 1 to 1.4 kg CO_2e/l and so three to four times higher than in the present study.

Our results seem to be in a reasonable range, but other studies have different locations, boundaries and features so that finding systematic differences is not really possible.

5. Conclusions

A life cycle assessment of traditional English beer brewed in a microbrewery has been made with blue water, cumulative energy and greenhouse gas emissions as used as measures of environmental performance. Cumulative energy demand was 4.4 MJ/l, GHGE were 0.38 kg CO_2e/l and blue water volumetric consumption was 3.4 l/l. This is up to delivery to the point of consumption.

The main burdens arise from raw material production, especially malted barley, brewing operations, delivery and with waste management relatively small but not negligible.

Other studies are not fully comparable, but these results are in the range reported for other beer types, countries and boundaries.

Opportunities for environmental performance are limited by the small scale of a microbrewery. More detailed auditing is needed to establish what could be achieved. Existing water consumption appears to be relatively low. Solar panels could help reduce the burdens of electricity use for temperature control.

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LCIA results of seven French arable crops produced within the public program AGRIBALYSE® - Contribution to better agricultural practices

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ABSTRACT

The program AGRIBALYSE® was an initiative launched by the French authorities (ADEME) to create a public Life Cycle Inventory (LCI) database of French agricultural products. The results and their uncertainties obtained for seven main French arable crops (soft wheat, maize, barley, rapeseed, sunflower, pea, and potato) are discussed here. A focus is made on four impact indicators: energy demand, GHG emissions, acidification, and eutrophication, while identifying the most contributing steps during crop life cycles. For the seven LCIs presented with the selected environmental impacts, the main contributors were field emissions. The assessment of practices such as organic fertilization or the introduction of intermediate crops could stress out some improvements on environmental impacts, but some impacts are opposed and emission models may be too simplified to simulate all their environmental benefits. These results represent a step forward to be shared with the agricultural sector in order to promote environmental evaluation and good farming practices.

Keywords: arable crops, LCI, national inventories, emission models, agricultural practices

1. Introduction

AGRIBALYSE launched in 2010 and finished in 2013, was a research program aiming to create a public database of Life Cycle Inventories (LCI) of French agricultural products. Among the 136 LCIs covering the main French agricultural production systems (arable and horticultural products, livestock products), this paper focuses on seven arable crops: maize, malting barley, rapeseed, soft wheat, spring pea, starch potato and sunflower.

This program had two goals: to produce agricultural LCIs for incorporation into the ADEME IMPACTS® database, in order to provide necessary information for environmental labelling, and to share data and methodologies to enable agricultural and food industries to assess their production chains and reduce their environmental impacts.

AGRIBALYSE datasets have several strengths. Firstly, LCIs of crops were constructed with a shared and harmonized methodology to ensure a consistent database and the comparability between LCIs (Koch and Salou 2013, Colomb et al 2014). This homogeneity is found for infrastructures and crop input. Specific adaptations were made for fertilizers, machinery, fuel consumption or input transport to correspond to French agricultural productions. Identical and adapted input data were used in all LCIs of crop productions. In addition to harmonized input data, pollutant flows are calculated with shared direct emission models best suited to the French context (Tailleur et al 2014). The allocation rules, between crops in cropping systems applied to all main crops, ensure the agronomic validity and strengthen the consistency of the database. The second asset of AGRIBALYSE is the data quality control. At different stages of the program, from data collection to Life Cycle Inventory Assessment (LCIA), experts and program partners validated data and intermediate results to ensure the quality of the AGRIBALYSE database.

The first aim of this paper is to present and discuss the LCIA results obtained for seven main French arable crops. The focus is on four impact indicators: energy demand, GHG emissions, acidification and eutrophication, while identifying the most contributing steps during crop life cycles. Emissions of phosphorus, nitrous oxide and ammonia calculated within AGRIBALYSE were compared to bibliographical references and experimental data. The second objective of this paper is to consider the use of these results by the agricultural sector. Improvements of cropping practices identified as a potential to lower environmental impacts were applied to these LCIs in order to estimate their efficiency.

2. Methods

Main methodological choices made in the AGRIBALYSE project for crop productions are summed up in Table 1. More detailed information is available in Koch and Salou 2013 and Tailleur et al 2014.

Table 1. Methodological choices for AGRIBALYSE crop productions

Products	Maize, malting barley, rapeseed, soft wheat, spring pea, starch potato and sunflower
Technology	Conventional production – average of farming practices in the different French regions
Time period	2005 to 2009
Boundaries	From cradle to farm gate - all up-stream processes (input production) are included but post-harvest operations (drying and storage) are excluded (for example maize LCI is expressed in kg of gross content with 28% of humidity).
Functional unit	AGRIBALYSE's functional unit is one kilogram of harvested product, in order to assess the environmental impacts of food products. However the functional unit used during data collection was one hectare of arable land and results per kilogram have been assessed using average yields. Both units are used in this article to better describe the multi-functionality of agriculture
Data collection	Mainly national statistics adjusted with experts' judgment.
Inputs data	Indirect emissions associated to crop inputs are based on existing data, mainly from the database Ecoinvent® V2 with many adaptations to the French context.
Direct emissions from polluting substances	Direct emissions were defined as flows of potentially polluting substances into the environment, directly associated with arable crop production on field: nitrous oxide (N_2O), carbon dioxide (CO_2) ammonia (NH_3), nitric oxide (NO), nitrate (NO_3^-), phosphorus (P) and phosphate (PO_4^{3-}), trace metals, soil loss, active substances, and land occupation and transformation. Changes in soil carbon stock were not taken into account in these results.
Allocations	Economic allocation between products / co-products (grain/straw). Animal systems carry the impacts of storage and processing organic manures and so they enter without burden in crop systems, except for a transport between farms. Direct emissions from manure application are allocated to crop productions. Allocation of fertilizers inputs, crop residues and nitrate leaching in crops sequences.
Data quality	Quality control was carried out during two phases of the program: at the end of the data collection process by French agronomic experts external to the AGRIBALYSE program and after the first calculations for the validation of calculated LCI and LCIA by the Technical Institutes as they did not carry out calculation. The Ecoinvent 2.0 pedigree matrix (Frischknecht et al, 2007) is used to assess the quality of individual inputs data and the quality of results of direct emissions models. In addition a global note is given to LCIs to fulfill ILCD requirements.

For impact assessment, no specific choice of midpoint indicators was made in the AGRIBALYSE project, its main goal being to develop a LCIs database; thus for each impact category several midpoint indicators were calculated. The data collection was relatively large to make sure of including the main impacts and methods presented in ADEME-AFNOR Working Group 1 (GT1) "Feed and fodder for domestic animals" or in the BPX 30-323 good practices guide (AFNOR, 2011). The indicators and methods selected for this communication (ILCD recommendation and CML) are presented in the Table 2.

The indicators aquatic ecotoxicity and human toxicity were not selected in this paper because there are large uncertainties in their calculation: the model to estimate direct emissions of pesticides is a rough method that quantified maximal emissions (100% of applied substances finish on the soil), only some active substances of pesticides are characterized in current characterization methods and the impact may strongly vary according to molecules used in the data collection. Methods are not mature enough to allow comparisons with references and to discriminate practices. Moreover, using national statistical data led to include a very large number of molecules in the LCIs. It does not reflect what is really happening in a specific field. Hence, a toxicity indicator using our national average LCIs would not reflect a real local impact on the environment.

Table 2. Selected midpoint indicators

Midpoint indicators	Direct flows and required data	Methods
Greenhouse gas emissions	Nitrous oxide (direct and indirect) and carbon dioxide	IPCC 2007 100 yrs - kg CO_2 eq
Energy consumption	Renewable and non-renewable energy	Cumulative Energy Demand (CED 1.8) - MJ
Acidification	Ammonia and nitric oxide	CML2001 - kg SO_2 eq
		Accumulated Exceedance (AE) - molc H^+ eq
Eutrophication	Substances involved in land and aquatic eutrophication (ammonia, nitrate, nitric oxide, phosphorus and phosphate)	CML baseline 2000 2.05 - kg eq PO_4^{3-}
	Substances involved in marine eutrophication (nitrogen flows : ammonia, nitrate and nitric oxide)	ReCiPe1.05H - kg N eq
	Substances involved in freshwater eutrophication (phosphorus flows : Phosphorus and phosphate)	ReCiPe1.05H - kg P eq

3. Results and discussion

3.1. Impact indicators

The main results of the chosen impact indicators are presented in Table 3. Results are expressed per hectare in order to compare the same agricultural land unit.

Table 3. Results of impact indicators for the seven studied crops, per surface unit (ha)

Impact indicators	Maize	Malting barley	Rapeseed	Soft wheat	Spring pea	Starch potato	Sunflower
Global warming - IPCC 2007 (kg CO_2 eq /ha)	3380	2480	2840	2820	940	3200	1210
Non renewable Energy (fossil + nuclear) CED 1.8 (MJ eq/ha)	27210	15490	15650	16720	9390	27480	8500
Acidification - CML 2001 (kg SO_2 eq/ha)	52.3	29.9	31.8	35.7	7.4	38.6	12.0
Acidification - Accumulated Exceedance, ILCD2011 (molc H^+ eq/ha)	92.6	51.8	55.0	62.2	11.2	64.7	19.9
Eutrophisation - CML baseline 2000 2.05 (kg PO_4^{3-} eq/ha)	29.5	22.4	24.2	25.2	20.6	25.2	18.7
Marine eutrophication - ReCiPe1.05H (kg N eq/ha)	37.1	33.6	38.2	37.9	37.5	39.3	27.9
Freshwater eutrophication - ReCiPe1.05H (kg P eq/ha)	1.8	1.5	1.1	1.3	1.2	1.3	1.7
Crop yield (kg raw material/ha)	10672	6618	3243	7100	4600	45780	2410

3.1.1. Greenhouse gas emissions (GHG)

GHG emissions range from 940 to 3380 kg eq CO_2/ha depending on the crop (Figure 1). In France, spring pea and sunflower have low global warming impacts because of no nitrogen fertilization on pea and an average of only 40 units of nitrogen per hectare for sunflower. When results are expressed in the functional unit, a kilogram of agricultural product, crops with low GHG emissions per hectare can have on average a bigger impact per kilogram of product or vice versa. Considering that 1 kg of rapeseed and 1 kg of potato, for example, have completely different characteristics in their water contents, energy contents and nutritional values, it could be difficult to compare their environmental impact per mass of product.

As already shown in previous studies (Gac et al. 2010, Carrouée et al. 2012, Nemecek et al 2014), fertilizers production and emissions of nitrous oxide (N_2O) are the main contributors of GHG emissions, representing from 63% (sunflower) to 81% (rapeseed) of the total GHG impact for crops with nitrogen fertilization in AGRIBALYSE results. The authors of the French program P-C-B[1] showed that mineral nitrogen production weighted from 29 to 75% of the total GHG impact and it reached from 52 to 83% of the total GHG impact if N_2O emis-

[1] CASDAR-7175 (“Pois-Colza-Blé” project) ‘Improvement of the economic and environmental performance of crop rotations with oilseed rape, wheat and peas’: the main objective was to assess and optimise the environmental and economic performance of crops’ rotations including oilseed rape and wheat, with and without grain legumes, with both field experiments and modeling studies

sions were added. Likewise the GES'TIM project, a French program to assess GHG impact and energy consumption of agricultural systems, led to the same conclusions, as for example 31% of the GHG impact of soft wheat is due to nitrogen production and 50% to N_2O emissions. As spring pea does not need nitrogen fertilization, crop residues account for most of the N_2O emissions (58%) with the method IPCC tier 1 whereas for the other crops, their crop residues represent only 15 to 31% of N_2O emissions and the rest of N_2O emissions is due to denitrification of the nitrogen fertilizers. In addition, machine use including fuel consumption has a higher GHG impact (41%) on spring pea than for other crops such as soft wheat where mechanization represents only 11%.

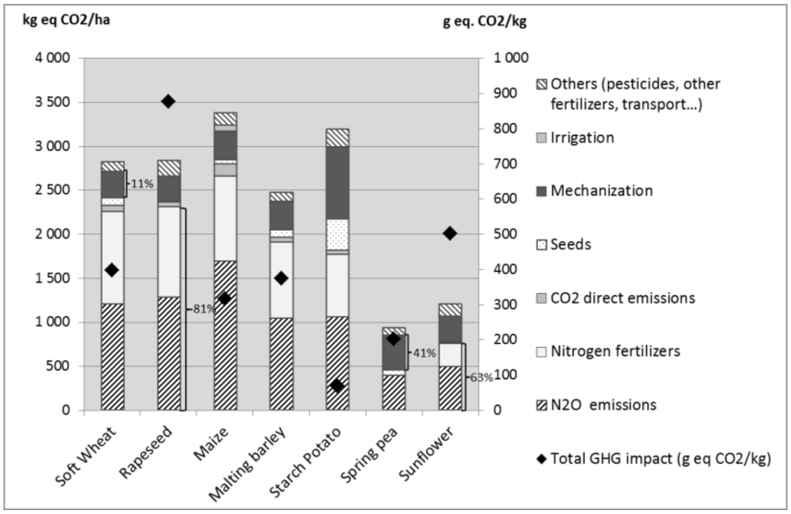

Figure 1. GHG impact of the seven studied crops per hectare (bars) and per kilogram of product (points)

3.1.2. Non-renewable energy demand

The non-renewable energy demand (fossil and nuclear) ranges from 8500 MJ/ha (sunflower) to 27480 MJ/ha (starch potato). The production of nitrogen fertilizers and mechanization are the main contributors to this impact. For soft wheat, rapeseed and malting barley, nitrogen fertilizers represent more than 50% of this impact (respectively 57%, 59% and 50%). Previous studies also showed that nitrogen fertilizers production requires a lot of energy and represents from 21 to 63% of the total energy demand (Carrouée et al. 2012, Nemecek et al 2014). Mechanization impacts the most on non-renewable energy consumption for starch potato, spring pea and sunflower (47%, 64% and 53%). In the GES'TIM project (Gac et al. 2010) mechanization for spring pea production represents 68% of the energy consumption. While this high contribution is explained by a comparatively low or

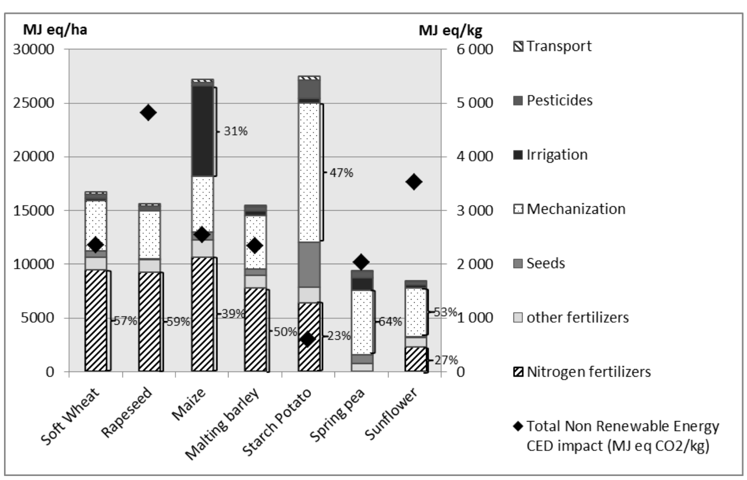

Figure 2. Non-renewable energy (fossil + nuclear) of the seven studied crops per hectare (bars) and per kilogram of product (points)

no mineral N fertilization for sunflower and pea, it is explained for starch potato by a high fossil energy demand for agricultural practices as soil management and harvest. For maize, the energy required for irrigation represents 31% of the impact.

3.1.3. Acidification

Acidification methods characterize fluxes of nitrogen and sulfur in the air, but direct emissions calculated for crop productions are only flows related to nitrogen (ammonia and nitric oxide), as SO_2 from combustion is included in mechanization. These direct emissions weight from 61% (starch potato) to 83% (maize) of the acidification indicator CML (Table 5), except for spring pea (31%) which receives no mineral nitrogen fertilizer. With the Accumulated Exceedance indicator, direct emissions represent a similar burden (67% to 87%) for crop systems.

3.1.4. Aquatic eutrophication

With the CML method, results range from 18.7 (sunflower) to 29.5 kg PO_4^{3-} eq/ha (maize). Direct nitrogen emissions (mainly NO_3^- leaching, but also NH_3 and NO flux) are the main contributors to the eutrophication impact (from 68% for sunflower to 81% for rapeseed of the total eutrophication impact), while phosphorus emissions weight from 9% (rapeseed and soft wheat) to 24% (sunflower). With the ReCiPe method, N and P emissions are separated in two midpoint indicators. Concerning marine eutrophication (N-dependent), impacts range from 27.9 (sunflower) to 39.3 kg N eq/ha (starch potato), the biggest impact being from nitrate emissions, 71% (potato) to 90% (sunflower) (table 5). For freshwater eutrophication (P-dependent), results span between 1.1 (rapeseed) to 1.8 kg P eq/ha (maize). Direct phosphorus emissions weight between 54% (soft wheat) to 79% (sunflower) of this total impact, mainly driven by phosphorus losses from soil erosion. The production of P fertilizers has also a significant impact on freshwater eutrophication, since it weights from 19% (sunflower) to 41% (soft wheat) of this impact. Regarding Ecoinvent references (Swiss crops) some AGRIBALYSE LCIs have the same freshwater eutrophication impact as soft wheat or malting barley. At the same time, some crops, like maize, have an impact nearly 50% higher than Ecoinvent data (due to higher direct phosphorus emissions).

3.1.5. Uncertainty analysis

An uncertainty analysis on AGRIBALYSE results shows that selected impact indicators are quite reliable, with coefficients of variation close to 10%. In Table 4 (uncertainty analysis on SimaPro®, 5000 runs), coefficients of variation for maize crop are presented as an example. Only eutrophication indicators with the ReCiPe method have a higher coefficient of variation (17.6 and 15.8% for marine and freshwater eutrophication).

Table 4. Coefficients of variation (CV) (%) of impact indicators for maize

CV (%)	Global warming	NR Energy	Acidification CML	Acidification AE	Eutrophication CML	Marine eutrophication ReCiPe	Freshwater eutrophication ReCiPe
Maize	10.5%	12.2%	9.1%	9%	9.9%	17.6%	15.8%

3.2. Field emissions impacting midpoint indicators

For crop productions, field emissions are the main contributors to the different presented environmental impact indicators except for energy demand (Table 5). Nevertheless, estimates of direct field emissions depend on the existence of valid models for a chosen scale. So, the AGRIBALYSE national scale has resulted in the choice of models that were valid for France and consistent with the available national data. Depending on the quantified substance, models can be really simple such as the quantification of N_2O with the IPCC method tier 1 or specifically developed to take into account the French environment as the nitrate leaching model (Tailleur et al 2014). Aware of the limits of models, some selected direct emissions have been studied and compared to French experimental data in order to identify possible ways of improvement for the future.

Table 5. Contribution of field emissions to impact indicators for the seven crops

Impact indicator	Main contributor flux	Range of value	Crops
Global warming - IPCC 2007 (kg CO2 eq)	Field emissions (N_2O, CO_2)	35 - 54%	All studied crops
	Nitrogen fertilizers production	21 - 37%	All studied crops (except pea)
Acidification - CML2001 (kg SO2 eq)	Direct field emissions	63 - 83%	All studied crops (except pea)
	Including NH_3 emissions	54 - 76%	
Marine eutrophication – ReCiPe 1.05H (kg N eq)	Direct field emissions	80 – 94%	All studied crops
	Including NO_3^- emissions	71 - 90%	
Freshwater eutrophication – ReCiPe 1.05H (kg P eq)	P emissions	54 – 79%	All studied crops

3.2.1. Nitrous oxide emissions

The NO GAS 2 project[2] collected experimental data of N_2O field emissions. 90% of the measured emissions in different parts of France vary from 0.2 to 2.3 kg N-N_2O/ha. Direct emissions calculated with the IPCC 2006 tier 1 method, are globally the same level (0.6 for spring pea to 3.1 kg N-N_2O/ha for maize). For two French crops, specific average emissions data are available from the NO GAS 2 project: 1.18 kg N-N_2O/ha/year for soft wheat (30 plots) and 1.08 kg N-N_2O/ha/year for rapeseed (34 plots). So AGRIBALYSE results tend to be over-estimated as calculated N_2O direct emissions of soft wheat and rapeseed crops are 1.9 to 2.2 times higher than measured emissions.

A sensitivity analysis was carried out to assess the impact of using these specific French references instead of the IPCC 2006 tier 1 method on the global warming impact of soft wheat and rapeseed. In the soft wheat and rapeseed AGRIBALYSE LCIs, calculated direct N_2O flux were replaced by the average measured emissions of N_2O from the NO GAS 2 project. The total GHG impact decreases with measured data from 2830 kg eq. CO_2/ha to 2250 kg eq. CO_2/ha for the soft wheat and from 2840 kg eq. CO_2/ha to 2220 kg eq. CO_2/ha for the rapeseed, representing a 20% reduction of the GHG impact.

As shown with the NO GAS 2 project, N_2O direct emissions are overestimated with the IPCC tier 1 method. In the short term, the ongoing development of a new equation tier 2 for France will allow a more accurate estimation of N_2O direct emissions and GHG impacts, given that N_2O is the major greenhouse gas from crop productions.

3.2.2. Phosphorus emissions

Phosphorus emissions represent 54 to 79% of the marine eutrophication impact (ReCiPe 1.05H). Few models exist to evaluate losses of phosphorus in surface water and ground water. The SALCA-P model (Nemecek et Kägi, 2007 and Prasuhn et al, 2006) was selected for its generic scope and its validity for all sources of emissions for major crops including grasslands. However this model was programmed for a Swiss situation, in particular the P-content of the soil, and it appeared that such data could not be adapted to the French context.

The SALCA-P model was used in the Ecoinvent datasets for crop productions and we compared Ecoinvent phosphorus fluxes with AGRIBALYSE's. SALCA-P estimates three fluxes: emissions of phosphorus from soil losses which reach rivers (soil losses were calculated with the RUSLE equation), emissions of phosphate in river by run-off and in groundwater by leaching. These three fluxes are turned into phosphorus for comparison and are showed in figure 3. On average, P losses in French agricultural systems vary between 0.6 and 1.1 kg P/ha whereas in Ecoinvent database, P losses are lower with values between 0.26 and 0.53 kg P/ha. There are no P losses by run-off with AGRIBALYSE assumptions, the slope being supposed to be 2% on average to restrain heavy metal

[2] NO GAS 2 project (still ongoing) aims to elaborate a new method to estimate direct N_2O annual emissions from cultivated soils (except pastures) based specifically on French experimental data. These data were collected through several national and European projects, from 2006 to 2012. The results presented here for soft wheat and rapeseed are the means of annual N_2O emissions measured on 30 and 34 plots for soft wheat and rapeseed respectively, located in Northern and South-West of France.

fluxes, and phosphorus in river from soil losses represents 90% of the direct P losses in the environment. P losses repartition is different in the Swiss references where run-off losses have the biggest contribution to P losses (45 to 64% of P losses).

Another important fact to point out is the difference between crops especially in AGRIBALYSE. The rapeseed, soft wheat and barley have close P emissions (around 0.6 kg P/ha) whereas spring pea and maize have higher losses (0.8 and 1.1 kg P/ha respectively). The difference is driven by losses of P from erosion and notably by crops factors used in the RUSLE equation. No specific data are actually available to validate these differences between crops.

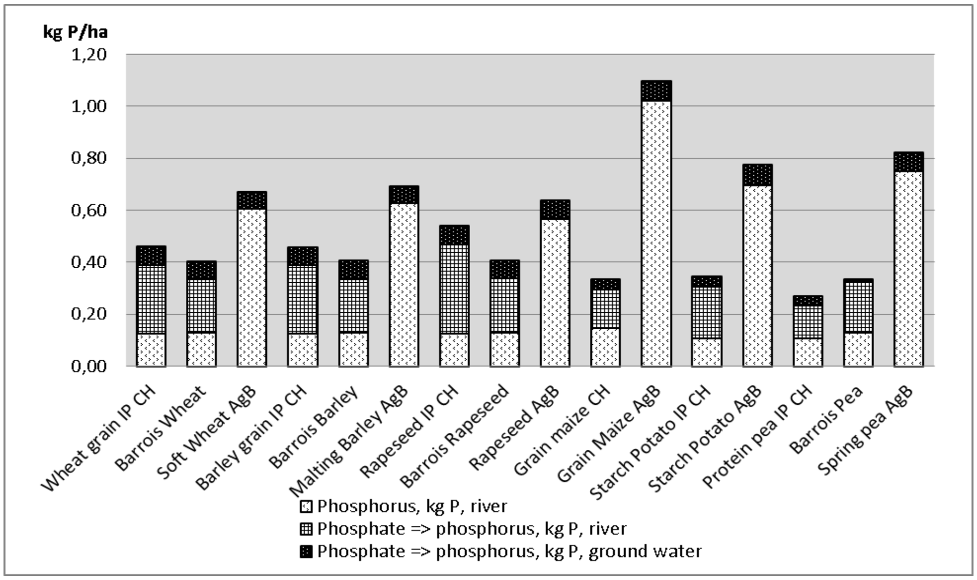

Figure 3. Comparison of phosphorus fluxes in AGRIBALYSE and Ecoinvent crops LCI. IP CH: Ecoinvent LCI for Swiss crops; Barrois: Ecoinvent LCI for a North-Eastern French region; AgB: AGRIBALYSE LCI

In addition, we compared calculated data with specific measured data from La Jaillière, an experimental site situated in the western part of France, where P losses have been measured for more than 10 years. Up to eleven plots have been studied with different agricultural practices and various privileged water circulations. Three selected plots with similar cropping systems are used for this comparison. They only differ from the water circulation in the plot: with or without drainage and with or without runoff. Two datasets were calculated using the SALCA-P model with the same agricultural practices: the first dataset was constructed with AGRIBALYSE asumptions for the model configuration and the second was configurated with La Jaillière specific data (for example, plot slope or P-content in the soil). The comparison of both datasets with experimental data brought out multiple results and outlooks.

First of all, P losses in characterization methods do not correspond to measured P fluxes in field experiments, which makes the comparison between experimental results and calculated data complicated. SALCA-P calculates emissions of phosphorus by erosion, and fluxes of phosphate by runoff and leaching whereas on experimental sites, losses by drainage and runoff are followed and data available for dissolved and particle phosphorus. Secondly, the SALCA-P model is not adapted for the La Jaillière site. When using accurate experimental data such as the soil phosphorus composition of La Jaillière, phosphorus fluxes calculated are lower than measured fluxes. Depending on the plot, measured data (on a 4-year period varying on average between 1.1 and 1.4 kg P/ha) are 2 to 7.6 times higher than calculated data (ranging from 0.2 to 0.6 kg P/ha) (Figure 4). This strong difference comes from the soil P-content and SALCA-P programmation. SALCA-P does not simulate all types of water circulation in a given field, it mainly represents orthonien runoff related to the slope and does not take into account hydromorph runoff linked to the water saturation of the soil. Eventually with AGRIBALYSE assumptions for SALCA-P configuration, total phosphorus losses are close to measured data in La Jaillière, even if the repartition between fluxes is completely different.

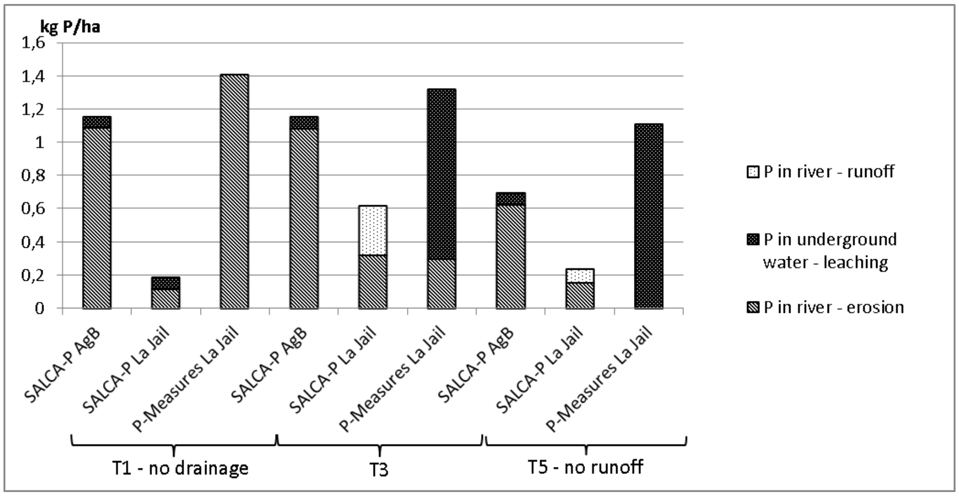

Figure 4. Comparison of phosphorus (2008-2011) fluxes in 3 plots of La Jaillière (T1, T3 and T5) – SALCA-P AgB: AGRIBALYSE assumptions; SALCA-P La Jail: specific site data; P-measures La Jail: experimental data

To conclude, AGRIBALYSE produced data near existing LCI references with the SALCA-P model. SALCA-P takes into account main phosphorus losses during crop productions but it is only partially adapted to the French context. SALCA-P is designed for a small scale such as an agricultural plot because it needs precise topographic data (slope, distance with a water body…) that are not available or too simplistic at a national scale. The choice of an average slope or the presence or absence of drainage at a large scale has a significant impact on phosphorus fluxes ; for example there is runoff of dissolved phosphorus in France even if factors used in AGRIBALYSE shows the opposite and precipitations (causing erosion and runoff) have an important impact on phosphorus losses (Breil et al. 2009). The absence of runoff with AGRIBALYSE assumptions limits also the sensitivity of the model to phosphorus nutrients, as mineral phosphorus inputs are only considered in losses of P by runoff. This work also shows that all phosphorus losses observed in France cannot be simulated by dynamic models. Few experimental data are available from French experimental stations. For example, P-losses were measured in the Parisot site in the south of France, between 1999 and 2007, and average P-losses by drainage were less than 0.1 kg P/ha. But it is only one more example that needs to be repeated. It means that further studies will be needed to gather experimental data, validate an average magnitude of P fluxes in the environment and develop models to assess phosphorus losses in relation to agricultural practices in the French context.

3.3. To identify good practices with LCA approach

The Life Cycle Assessment, a multi-criteria evaluation, allows a holistic approach of any production. It can be used to measure the environmental impact of some proposed agricultural practices while controlling that environmental fluxes are not transferred to other environmental compartments. One of the goals of the AGRIBALYSE's project is to help eco-design of food products, providing a methodological framework to assess impacts of production systems and first average references. Also AGRIBALYSE results should allow us to evaluate possible improvements of farming practices in the future.

Among good agricultural practices, introducing a leguminous crop in the rotation can be highlighted with AGRIBALYSE results. The analysis of crops LCIA confirms the significantly lower potential impacts of pea, a N-fixing crop, or of a low input crop such as sunflower, compared to crops requiring a significant amount of nitrogen fertilization. This was already underlined by Nemecek et al (2014) both at crop level and at rotation level and confirmed by field experiments (Jeuffroy et al. 2013) which enabled to estimate a decrease in N_2O field emissions by 20–25% when introducing a pea crop in a three-year rotation.

Here a specific work was done to evaluate the sensibility of AGIBALYSE methodology to farming practices with the substitution of some nitrogen mineral fertilizers by organic manure. For this simulation, boundaries of the LCA were extended from a single crop to a whole crop rotation with rapeseed, soft wheat and spring barley, regarding the fact that fertilization and especially organic fertilization is thought at a cropping sequence scale. Apart from that, choices for allocation and emission models were from AGRIBALYSE. When fertilizing an agricultural plot with organic manure, only a part of the organic nitrogen will be available for the crop depending on the type of manure (pig slurry, cattle manure, compost…), the period of application and the receiving crop. We compared a scenario with only mineral fertilization, based on data collected within the AGRIBALYSE project, to a scenario designed with pig slurry spreading. This scenario was built to apply the same amount of nitrogen as in the mineral scenario using equivalence factors for available organic nitrogen. It was also important to manage the quantity of total phosphorus brought to the plot with animal slurry. We supposed that the soil had a medium P-content and that the farmer wanted to keep this balance. So the total amount of phosphorus fertilizer (mineral and organic) is equal to the total P-exports of the rotation crop products. In values, 175 kg of mineral nitrogen are spread on rapeseed in both scenarios, respectively 164 and 133 kg of mineral nitrogen fertilizer on soft wheat and spring barley for the mineral scenario, and 121 and 85 kg N mineral/ha for the manure scenario with a total organic manure of 130 kg N/ha shared between the soft wheat and the spring barley. All other inputs and agricultural practices are AGRIBALYSE data.

LCA were calculated for both systems to compare the environmental impact of a chosen fertilization. Depending on the midpoint indicator, both systems did not evolve in the same way. Regarding GHG impact, there was no significant difference between both scenarios: the substitution of mineral fertilizers by pig slurry resulted in the diminution of the impact of nitrogen production but increased indirect nitrous oxide from airborne deposition of volatized nitrogen. The EMEP/EAA method tier 2 estimates important quantities of NH_3 from the organic fertilization (40% of the N-NH_4^+ brought from pig slurry is estimated to be emitted to the environment). In terms of energy, the scenario with organic nitrogen needs less non-renewable energy and decreases from 16460 MJ/ha/year to 13700 MJ/ha/year for the rotation, representing a diminution of 17% of the non- renewable energy use. With those two indicators the scenario with organic manure tends to have better environmental impacts than the scenario with only mineral nitrogen. Only when analyzing the acidification impact, because of important NH_3 emissions linked with manure fertilization, the all-mineral scenario has an acidification impact 40% lower than the scenario with pig slurry.

This scenario comparison brings out many challenges. When using a multi-criteria approach, the difficulty is to weight the different impact indicators to be able to correctly judge the different practices. In the analyzed case for example, organic fertilization has a better impact on non-renewable energy, but a worst impact on acidification than the scenario with only mineral nitrogen. No conclusion can be highlighted without defining which of energy use or acidification is the most important criteria. As the selected emission models do not take into account all agricultural practices, only few agricultural practices recommended as beneficial to the environment, such as replacing mineral fertilizers with organic manure, introducing intercrops or increasing the length of the rotation with protein crops, could be identified and tested with an LCA approach. When spreading organic manure, NH_3 emissions can be reduced if manure is incorporated to the soil after application (Sommer et Hutchings 2001). The EMEP/EAA method does not differentiate the mode of manure application or post-application practices, and tends to overestimate NH_3 emissions. We used experimental data to approach the potential reduction of incorporating the pig slurry after its spreading. Data collected in the site of Bignan - West of France – showed that only 11% of the N-NH_4^+ was emitted as N-NH_3 if the pig slurry was incorporated after spreading (Cohan et al. 2013). This reference used in the pig slurry scenario decreased by 36% the total acidification impact of the organic scenario and in that case both scenarios have the same level of potential acidification on the environment (respectively 86.9 kg SO_2 and 91.7 kg SO_2 eq/ha/yr for the mineral scenario and the pig slurry scenario).

Eco-design and practices assessment require accurate emission models and detailed agricultural data which are not always compatible with a national scale. In this context, AGRIBALYSE methodological choices correspond to a compromise between national database and accurate emission models.

4. Conclusion

With the AGRIBALYSE project, the most relevant choices were made to evaluate French agricultural productions on a national level and to produce valid LCI references to supply the ADEME IMPACTS® database, thus providing necessary information for environmental labelling. This first analysis of AGRIBALYSE LCIA validated analyzed LCIs and has to be continued especially for the other crops (durum wheat, alfalfa, etc.). Crop LCIs meet the need for harmonized data and give keys to evaluate environmental impacts of agricultural practices; it is a first step in using LCA for eco-design in agriculture even if the complexity of LCA (which requires large quantities of data) is a constraint for its application to agronomy. Nevertheless, remaining work needs to be done to reduce the lack of simplified emission models and to develop agricultural practices more accurately.

AGRIBALYSE LCIs of arable crops should be updated with regard to current input data on one hand (changes in agronomic practices, production of inputs, …) and on the other hand in respect of the evolving methods and models for calculating emissions (new EMEP methodology tier 2 was released in 2013 for mineral fertilizers, ongoing work on a tier 2 equation for direct N_2O emissions) which could take the specific characteristics of crops, agricultural practices and soil-climate characteristics better into account. Further developments will be needed for some flows such as the dynamics of carbon in the soil or the fate of pesticides after their application, and some environmental impacts, as biodiversity or water consumption. Finally, these results represent a step forward for the calculation of French agricultural systems LCA to share with the agricultural sector in order to promote environmental evaluation and good farming practices. LCA provides tools to weight different potential impacts and to compare different scenarios while identifying major damaging activities of an agricultural product or a production system. That is why LCA is complementary to other environmental assessment methods (assessment of farms, local scale, impact studies, etc.) and will help to improve agricultural practices.

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Swine Logistics Assessment post-farm to market

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ABSTRACT

Brazil is one of the most important producers, exporters and consumers of swine in the world. Among all states, Santa Catarina stands out in the national scenario. The main producing area in this state is about 500 km away from the nearest seaport, so that the logistic involves two steps: (a) live weight transportation to the slaughterhouse, and (b) frozen carcass through refrigerated transportation to market. Those features influence the environmental performance of the final product. We applied the Life Cycle Assessment methodology to analyze five different scenarios performing carbon footprint (CF) and energy demand (ED), varying slaughterhouse location and thus distances of cooling and live transportation. In general, results showed that lower live weight distances would represent lower impacts on CF and ED, although it is possible to reach an optimum mortality rate that would invert scenarios rankings.

Keywords: LCA; Logistics; Post-farm, Swine, Brazil

1. Introduction

Brazilian swine production is an important activity, with a herd of 35 million heads, representing the fourth largest producer (3 millions ton/year), fourth largest exporter (600 thousand ton/year) and the sixth largest consumer (11–13 kg/inhabitant/year) in the World (Kunz et al. 2009; USDA 2013). In this context, Santa Catarina state (southern Brazil) is the largest producer in the country, with 19.3% of the national herd (IBGE 2012).

The swine production chain in Santa Catarina can be described in 5 major steps, commonly known as piglet production, weaning to finishing, slaughtering (meat processing), market and consumption. Several Life Cycle Assessment (LCA) studies concluded that its main environmental hotspot lies in the livestock processes (Basset-Mens; van der Werf 2005; Dalgaard, 2007; Elferink et al. 2008; Eriksson et al. 2005; Kingston et al. 2009; Kool et al. 2009; Nguyen et al. 2011; Wiedemann et al. 2010) due feed production and animals emissions. With reflection, this step has been widely studied in order to reduce impacts.

Nonetheless, there are other production processes that can be assessed in order to improve pork's ecoprofile, or indicate preferable paths inside its production chain. For instance, after the weaning-to-finishing process, swine have two logistic steps before reaching the final consumer: (a) liveweight transportation to the slaughterhouse, and (b) frozen carcass transport through refrigerated transportation to market (internal or external).

Food transport refrigeration is a critical link in the food chain not only in terms of maintaining the temperature integrity of the transported products but also its impact on energy consumption and CO_2 emissions (Tassou et al. 2009). In these processes mechanical refrigeration technologies are invariably employed and they contribute significantly to the environmental impacts of the food sector (Tassou et al. 2010).

The impact of this technology varies depending on the distances and types of transportation involved. Those features also influence the environmental performance of the food final product. Transportation type (a) can have lower environmental impacts than type (b), since refrigerated transportation requires more inputs (e.g. fuel and refrigerant gases). However, long distances for live weight transportation are related with animal's welfare reduction, what can bring high mortality for the swine (up to 1%) (Alverós et al. 2008; Gonsalvez et al. 2006; Vecerek et al. 2006).

When it occurs, swine loss (during handling and transport) is usually preceded by a period of poor welfare (European Commission, 2002). Potential factors influencing this issue have been compiled (Adams, 1994; European Commission, 2002; Broom, 2005; Schwartzkopf-Genswein et al., 2012) and strategies have been developed to provide better welfare (European Commission, 2002; Broom, 2005; Silveira, 2013).

Brazilian laws are out of date (Silva et al. 2009) and are more focused on breeding and slaughtering processes (Brasil, 1952; 2000; 2008; 2011). Although, EMBRAPA and Brazilian Association of Swine Breeders have created a Manual of Good Agricultural and Livestock Practices in Swine Production (2011).

Since meat products have its main environmental hotspot in the livestock processes, any loss on finished swine represents higher environmental impacts.

Therefore, the aim of this study was to find the best combination on swine logistics post finishing to market. We applied LCA methodology to compare possible scenarios for Santa Catarina state, an important swine producer in Brazil. Seeking for lower Carbon Footprints (CF) and Energy Demand (ED), we evaluated five different scenarios to deliver frozen swine to São Francisco do Sul's City Seaport, 500 km away from the producer city, Concórdia.

2. Methods

Lived swine on conventional transportation (normal truck) to the slaughterhouse, and then refrigerated transportation to the harbor (for export) or storage (for national consumption), was varied in five scenarios, as shown in Table 1.

The Functional Unit (FU) was 10 tons of refrigerated swine delivered at the seaport. Therefore, the reference flow was 10 tons of frozen carcass in refrigerated transportation and 13.5 tons in live weight transportation.

Scenario 01(SC 01) was designed to represent a swine that is entirely transported frozen until the seaport. In this case, the slaughterhouse is hypothetically located beside the swine producers, therefore there is need for only a 50 km range of live weight transportation, from producers to the meat processing industry. This scenario featured a transportation with minor live swine losses due the lower distance attributed to the live weight transport, although represents a higher diesel consumption for refrigerated transport and other inputs and outputs such as refrigerant gases, due to cooling.

Scenario 05 (SC 05) is the opposite, with meat processing near to the seaport, and all the transport by regular trucks until the slaughterhouse. In this case, live transport does imply in meat losses, but on the other hand, fuel consumption is lower.

Scenario 03 (SC 03) is equalized with the same distances between producers, slaughterhouse and the seaport. Thus was modeled in 250 km for both kinds of trucks. Scenarios 02 (SC 02) and 04 (SC 04) were designed to have the slaughterhouse near the producers or near the seaport respectively, in a range of 100 km, varying transportations types.

Other characteristic of live weight transportation is that it carries more mass due to inedible offal. Although some parts are allocated after slaughtering, it means 35% more ton.kilometers^{-1} compared to frozen transportation. This difference is equalized by reference flow.

Table 1. Distances for modeled scenarios.

Scenario	Live weight transportation	Frozen transportation
SC 01	50 km	450 km
SC 02	100 km	400 km
SC 03	250 km	250 km
SC 04	400 km	100 km
SC 05	450 km	50 km

To create the inventory, primary data were collected from an Agroindustry for trucks, both refrigerated and regular, diesel consumption, truck type and capacities, live and deadweights, transport losses and city locations. The swine mortality accounting was from Cherubini et al. (unpublished data). For welfare conditions and swine losses rates, see Alvaróz et al., (2008); and table 2 based on not fasted pigs (Alvaróz et al., 2008) respectively. Other data was collected from literature and Ecoinvent Database® (for refrigerant gases). Data from cooling machinery production were not included.

Table 2. Life Cycle Inventory.

Scenario	Live cargo transportation	Amount	Frozen cargo transportation	Amount
All	Transport, lorry 16-32 ton/Euro3/EUR R	Table 1	Adapted from Transport, lorry 16-32 ton/Euro3/EUR R (135% higher diesel consumption**; Refrigerant gas[1])	Table 1
SC 01	Swine Losses*	33.7 kg	-	-
SC 02	Swine Losses*	43.2 kg	-	-
SC 03	Swine Losses*	54.0 kg	-	-
SC 04	Swine Losses*	70.2 kg	-	-
SC 05	Swine Losses*	80.8 kg	-	-

* Alvaróz et al (2008)
**Tassou et al. (2009)
[1] Personal communication, from south Brazilian agroindustry.

The Life Cycle Impact Assessment was performed with the software SimaPro® 8.0 using the CML-IA method for CF and the Cumulative Energy Demand method for ED.

3. Results and Discussion

Results have shown better environmental performance to the SC 01 with the shortest distance between producer and slaughterhouse (50 km), despite of the long distance covered by refrigerated transport (450 km). This scenario had 1.07 ton CO_2 eq. and 18,268.05 MJ eq. (Table 3), values 27% and 15% lower than the worst scenario (SC 05), with the slaughterhouse near the seaport (Figure 1). These emissions represent approximately 3% of the total cradle-to-gate swine production chain, based on Dalgaard (2007).

Table 3. Life Cycle Impact Assessment.

Impact category	Unit	SC 01	SC 02	SC 03	SC 04	SC 05
Global warming (GWP100)	ton. CO_2 eq.	1.07	1.13	1.25	1.39	1.46
Total cumulative energy demand	MJ eq.	18,268.05	18,745.15	19,883.33	21,111.71	21,611.36

The other scenarios are between SC 01 and SC 05 for both impact categories, with SC 02 representing lower impact than SC 03 that represents lower impacts than SC 04.

As it can be seen in Figure 1, SC 02 is 4% higher on CF and 2% higher on ED than SC 01, while SC 03 is 9% and 5% higher than SC 02 and SC 04 also presents this pattern with values 10% and 6% higher than the last scenario.

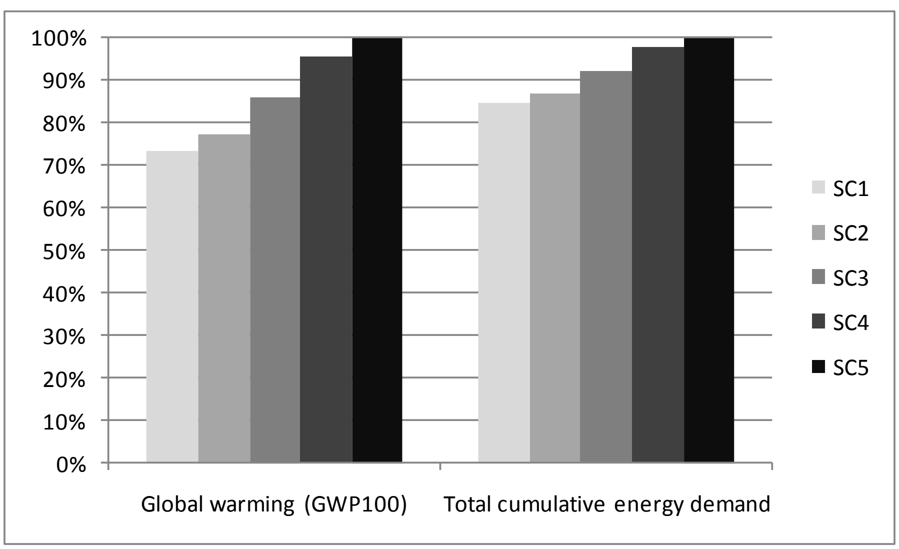

Figure 1. Environment impact comparison.

The difference in performance between lower and higher scenarios (SC 01 and SC 05) occurs due to swine losses during the long live weight transportation and also due to higher mass transported per km, even allocating slaughterer byproducts (i.e. offal). Swine losses means that impacts from feed production and animal rearing are attributed to live weight transport.

Analyzing extreme scenarios (SC 01 and SC 05) without live swine losses, we found that the CF is still higher for live weight transport, i.e., SC 01 remains better. On the other hand, for ED, SC 05 becomes preferable to SC 01. On top of that, it is possible to indicate an optimum rate of mortality where live weight transport could be the best choice.

5. Conclusion

This paper performed a LCA of swine logistics scenarios post-farm to market, varying conventional transportation (live weight) and frozen carcass transportation (cooling trucks) distances, with different inputs and outputs. Scenarios were based on the Santa Catarina state reality, from Concórdia until São Francisco do Sul, with support from an Agroindustry for providing primary data.

Locating the slaughterhouse as near as possible to producers would reduce the impacts around 27% and 15% for CF and ED, respectively, if compared to locate the meat processes near final destination (i.e. seaport for exportation).

It is clear that for CF, the reduction of live weight transportation yields better environmental performance. This statement is also true for ED results (for scenarios assessed). Although for ED, decreasing swine losses in live weight transport would allow one to reach lower energy consumption than frozen transport. Hence, it is possible to reduce mortality rates to invert scenarios' rankings. Nevertheless, the first step to reach a better environmental performance is to reduce swine losses on transportation.

These results lead us to a conclusion that they are in accordance with the recent reports on livestock as well as recent laws, where animal welfare provision is recommended and required. Besides its ethical side and influence on meat quality, animal's welfare can lead to a decrease on potential environmental loads through swine's death reduction.

Other environmental impact categories would be relevant to evaluate the overall environmental performance. For instance, frozen transportation could be a significant source for toxicity and ozone layer depletion impact categories. There is lack of cooling machinery data in this study, but it would not change the results significantly because the machinery lifespan is long and could be used for many FU, such that its impacts would be diluted.

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